
Evaluation of Techniques for
Estimating Fishery Assessment
Parameters in the Tasmanian Rock
Lobster Fishery

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A handwritten signature in cursive script, reading "Stewart Frusher". The ink is dark and the handwriting is fluid.

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Abstract

Rock lobsters are one of the premiere seafood products around the world. High demand has led to most lobster fisheries being over or fully exploited. The Tasmanian rock lobster fishery is no exception and has become a major industry for Tasmania since its rapid commercialisation in the early part of the last century. The Tasmanian fishery, based on the southern rock lobster *Jasus edwardsii*, is the backbone of the Tasmanian fishing fleet and provides valuable socio-economic input into many of Tasmania's coastal rural towns. For this reason, the Government requires scientists to try to provide accurate and precise assessments of this fishery for their managers.

The most recent change in the assessment of this fishery was the development of a mathematical assessment model. In addition to assessing the current state of the resource, the model has forward projection capabilities so that future harvest strategies can be evaluated. Like all fishery models, the Tasmanian assessment model is based on a number of assumptions for estimating biomass and egg production. In addition, the model assumes that the dynamics of fishing remain constant from year to year. However, the dynamics of the fishery are changing as management, technology and markets change the behaviour of fishers. The change to an Individual Transferable Quota management system in 1998 has seen fishers focus on the dollar return per kilogram, rather than maximising their catch, as a way to improve profitability. Global positioning and echo sounder technology enable fishers to locate and chart lobster habitat better than ever before, and the rapid expansion of air transportation has seen the opening of Asian markets for premium priced live lobsters. To ensure that model estimates are reliable under changing patterns of exploitation, model estimates need to be validated. This is best achieved by estimating the same parameters using different techniques.

Trials to obtain estimates of exploitation rate and biomass using change-in-ratio (CIR) and index-removal (IR) techniques have provided encouraging results for southern regions of the Tasmanian rock lobster fishery. The latter two methods require that there be (at least) two surveys within a year, with harvest(s) occurring between surveys. The two methods have generally provided similar results. However, on

occasion, the CIR and IR results were widely divergent with the IR estimates of exploitation rate much higher than the CIR estimate. I examined the assumptions required to be met for use of each of these techniques, especially the assumptions regarding catchability. Diagnostic tests were developed for each of the techniques to ensure that the assumptions of catchability are met. The CIR technique, which has a weaker assumption of catchability, was more robust than the IR technique. Application of the diagnostic tests resulted in several estimates being discarded. Despite this, exploitation rate estimates were available for five of the six fishing years. The diagnostic tests also demonstrated when an earlier than expected moult had occurred in the fishery. This moult affected the end of season sample which could no longer be used to obtain exploitation rate estimates.

In northern regions of Tasmania, moulting occurs within the fishing season and the CIR and IR techniques can not be used. This thesis evaluates an alternative approach to stock assessment using multi-year tagging studies to estimate fishing and natural mortality in northern regions of the fishery. Data obtained from a tagging project undertaken from 1992 to 1995 were analyzed. The most parsimonious model was based on using three tagging events each fishing season, and estimating annual fishing mortalities and a single natural mortality estimate over the duration of the study. Fishing mortality was partitioned to the period of the year based on the amount of fishing effort between tagging events. Natural mortality was partitioned to the period based on the amount of time elapsed between tagging events. Although annual fishing mortalities could be estimated for each sex, a more parsimonious model was obtained when female fishing mortality was set as a proportion of male fishing mortality dependent on the amount of fishing effort in the female fishing season relative to the amount of effort in the entire fishing year. Tag reporting rate was also held constant in the model over the period of the study. Results demonstrated that relatively precise estimates of annual fishing mortality and tag reporting rate could be obtained but natural mortality was imprecisely estimated. Annual estimates of instantaneous fishing mortality were high, averaging around 1.0 to 1.2 per year, and were similar to those obtained by the assessment model. The precision of annual fishing mortalities estimated in the years after tagging ceased declined due to the low number of tags returned. Low tag returns were associated with the high exploitation rates and low tag reporting rate (estimated to be 22%). Natural mortality was estimated for all years

combined. The estimate of natural mortality was zero with a standard error of 0.2 per year. Natural mortality is an extremely difficult parameter to estimate. Current estimates, which are used in models for southern rock lobster throughout its range in Australia and New Zealand, are based on a small number of long term recaptures from southern Tasmania.

Often, in fisheries such as the Tasmanian rock lobster fishery, a major management objective is to rebuild the stock and lower exploitation rates. Under declining exploitation rate, the need for a precise estimate of natural mortality increases. This thesis investigated four different ways to improve precision of estimated parameters using multi-year tagging models. Simulations were patterned after that Tasmanian rock lobster fishery and showed that the best gains in precision were obtained by either increasing the tag reporting rate or increasing the duration of the study. Although there was considerable potential to increase tag reporting rate as the estimate from the above study was low, there can be no guarantee that either increased rewards or improved publicity will result in an increase in tag reporting rate. The most certain way of increasing the precision of natural mortality was by increasing the duration of the study. This thesis suggests a design based on three years of twice a year tagging followed by three years of once a year tagging.

This thesis also investigated selectivity estimates from the fishery because selectivity is assumed to be constant from year to year in the assessment model. Selectivity was found to change as the size composition of the lobster population changed. Large lobsters were found to inhibit the catchability of small lobsters. As large lobsters are removed from the population over time the catch rate of small lobsters can be expected to increase. Thus selectivity can be expected to vary as a function of the exploitation rate. Assessment models, which are based on size structure, need to account for changes in selectivity as the size structure of the population changes. Otherwise, if the increase in catchability of small lobsters is not accounted for, it is likely that declining recruitment to the fishery will not be fully detected.

This study found selectivity to be an important parameter in some crustacean trap fisheries and identified the need to validate the assumption of constant selectivity both within and between fishing seasons.

Acknowledgements

PhD theses generally require a dedicated effort and single mindedness that impact on those close to the candidate. However, when the candidate has additional pressure through existing work requirements and family commitments then the impact on work colleagues, friends and family is exacerbated.

I acknowledge both the Director of the Tasmanian Aquaculture and Fisheries Institute, Professor Colin Buxton and the Dean of the Faculty of Science and Engineering, Professor Paul Haddad who supported my six month study leave to the Virginia Institute of Marine Science where I worked with my main supervisor Professor John Hoenig. I also thank Professor Buxton who supported my combined work and study chaos after arriving back from Virginia. Dr Caleb Gardner's who ably led the Crustacean section during my absence and, in an unofficial capacity, maintained a leadership perspective on my return. My colleagues in the Crustacean section also put up with 'weird, tired and odd' behaviour during this time.

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The brunt of the impact of undertaking and completing the thesis was borne by my family, particularly my wife. Undertaking all the additional duties in an already heavily committed household was a level of support that can never be repaid. I will always be deeply in debt.

Finally, my children persevered with the added level of chaos placed on their lives. However, there was one occasion when I knew that there could be no turning back:

On one hot and humid summer morning in Virginia I had hassled my daughters on several occasions to hurry up so that I could arrive at work early. Having driven to the end of driveway I suddenly remember that I had left my notes behind. My wife commented that I was 'becoming an absent-minded professor like John' (my supervisor). My daughter quickly replied "Dad's not an absent-minded professor, he's just an absent-minded person without a PhD". Something **had** to be done!

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1. Brief history of the development and management of the Tasmanian southern rock lobster fishery.

The southern rock lobster, *Jasus edwardsii* (Hutton) is a member of the spiny lobster group belonging to the family Palinuridae. It has a broad distribution across Australasia, being found in southern Western Australia, South Australia, Victoria, Tasmania, southern New South Wales, and New Zealand (Winstanley 1977; Annala 1983).

Major fisheries for *J. edwardsii* occur in South Australia, Victoria, Tasmania and New Zealand (Kailola et al., 1993). In Australia, the annual landed catch is approaching 6,000 tonnes and is valued at approximately 150 million Australian dollars making it one of Australia's more valuable fisheries. Lobsters are also caught recreationally by diving and trapping with over 25,000 recreational licenses issued annually in southern Australia.

Management of this important resource is based on an annual stock assessment (Frusher, 1997). This dissertation addresses methods used in assessing the state of the rock lobster resource in Tasmania.

1.1 Social and Economic Importance

Rock lobsters in Tasmania were an important source of food for coastal aboriginal tribes and this was also the case for the first European settlers, who arrived in Hobart, Tasmania in 1804. While the aboriginal population valued rock lobsters for its food and barter value, Europeans quickly began to appreciate its commercial potential. In 1882, during a Royal Commission inquiry into the State of Tasmania's fisheries, Seals et al. (1883) wrote

*“The crayfish (*P. edwardsii*) [*J. edwardsii* was known as *Panulirus edwardsii* in 1882] is, perhaps, one of the most important of our marine products, being not only esteemed for its quality, but for its great commercial value from its wonderful abundance, especially around our eastern coasts”.*

The fishery continued as a major economic activity and Winstanley (1973) described the fishery from 1904 to 1972 by saying:

“The Tasmanian fishing industry has been built around the development and stability of the rock lobster fishery and the majority of men working in the primary and secondary phases of the industry have been directly dependent on the prosperity of this fishery for their livelihood.”

The southern rock lobster continues to support one of Tasmania’s most important wild harvest fisheries. The annual catch from the commercial rock lobster fishery is valued at approx. \$45 to \$50 million. The rock lobster fishing industry is still considered the backbone of Tasmania’s fishing fleet with over 250 vessels, the majority working out of Tasmania’s coastal rural towns. The industry directly employs about 761 people in Tasmania (Williamson, Wood and Bradshaw, 1998) and contributes significant socio-economic benefits to many of Tasmania’s coastal regions. Over 80% of the licenses are held by Tasmanians, with the majority being owner-operators. The industry spends between 24 to 36% of the landed value of the catch on materials and approximately 41% on labour, thus being a valuable contributor to regional employment and economic activity. The processing sector is dependent on live holding facilities as approximately 74% of the catch is marketed live. The rock lobster processing sector is highly specialised adding to the socio-economic benefits that the rock lobster industry contributes to Tasmania.

1.2 History

1.2.1 Catch Rates

The southern rock lobster was bountiful prior to commercial exploitation and its vulnerability to baited rings and traps supported emerging commercial fisheries. Kerr (1985) records catches as high as 2,400 lobsters in a day using cray rings (a baited hoop net). In the 1920's, after the legalisation of traps as a fishing method (Winstanley, 1973), catch rates of around 30 lobsters per trap lift were considered common (Kerr, 1985).

Historically, the commercial fishery developed around the established towns on the weather protected east coast in the late 1800's and early 1900's. As catch declined in these regions and technology improved, vessels moved to deeper waters and to lesser-protected waters of the west coast. In 1997 nearly 70% of the catch came from the west coast.

1.2.2 Technology

From its humble beginnings where sail and oar powered vessels set out from Hobart Town (Seal et al. 1883), the fishing industry has embraced technological improvements in its relentless pursuit of lobsters. Diesel motors quickly replaced sail power and, after the second world war, mechanical trap haulers, refrigeration units and pumps, synthetic ropes and buoys, echo sounders and, more recently, global positioning systems all increased the efficiency of the rock lobster fishing vessel (Smith and Ferguson, 1969).

Technology has also played an important role in the development of markets for rock lobster. Prior to the Second World War, rock lobsters were primarily sold in local markets in Hobart and Melbourne. The cost effectiveness and availability of refrigeration after the Second World War provided the opportunity to ship frozen product to overseas markets. In particular, the American frozen tail market created a boom in the Industry in the 1950's and 1960's (Smith and Ferguson, 1969). The improved efficiency and reduced cost of air transport enabled the live export trade into Asian markets. The higher prices obtained for the live product has resulted in Asian markets dominating lobster exports in the 1990's.

1.2.3 Management

The lobster's culinary position as a high priced delicacy has contributed to its decline. As early as 1882 the "Royal Commission on the Fisheries of Tasmania" heard Hobart fishers' concerns regarding the state of the local stocks

" . . . The destruction of crayfish so serious in some localities as to threaten extermination at no distant date "

and two years later Saville-Kent (1884) wrote

" . . . Such a deterioration has in fact, I understand, already begun to make itself felt, the fish [lobsters] now captured being less in both numbers and dimensions as compared with former years".

The concerns resulted in the first set of management regulations introduced in 1889. Regulations included a minimum legal size and a prohibition on taking soft shelled (recently moulted) lobsters or berried female lobsters. The taking of soft shelled lobsters was replaced by a closed season in 1947. Although there have been variations to these regulations, they have remained in place and are enforced as part of the current quota management system introduced in 1998.

The early concerns regarding a rapid demise of the rock lobster stocks proved to be unfounded and with the management controls mentioned above, the fishery saw increasing annual fishing effort and efficiency result in increased catches up until the 1960's. During the 1960's the catches showed signs of leveling off while effort continued to increase. This resulted in the maximum number of licences being fixed at 420 in 1967 and in 1972 the maximum number of traps allowed in the fishery was set

at 10,000 (Winstanley, 1973). In particular, the eastern half of the State of Tasmania show only a limited increase in catch from 1946 to 1970 despite a 300% increase in the number of traps licensed, a 75% increase in the number of vessels licensed, a reduction in the minimum size limit of female rock lobsters and substantial technological advancements in fishing vessels and equipment (Winstanley, 1973).

The impact of the fishing pressure on the eastern and southeastern Tasmanian lobster stocks was also reflected in the size structure of the landings. Winstanley (1973) found that the annual yield from these regions was primarily dependent on annual recruitment from undersized stocks. Frusher (1997) found that male recruits comprised 80% and 76% of the male portion of the fishery in the southeast and eastern regions of Tasmania in the 1990's, respectively.

In addition to the commercial fishery, Tasmanians have a long tradition of recreationally fishing for rock lobsters (often referred to as crayfish) and many Tasmanians associate summer holidays with recreationally caught lobster. Winstanley (1973) states that

“There is probably no other state in Australia where rock lobster have for so long attracted the attention of so many amateur fishermen or where the general public has been more aware of the extent and economic worth of the rock lobster resource”

The recreational fishery was also not immune to the decline in abundance of rock lobsters. In 1951 there was a move to ban the use of traps by amateur fishermen in an effort to protect the inshore stocks. Although this was defeated, in 1956 the spearing of rock lobster was banned and amateur fishermen were restricted to the use of one trap in an effort to protect the ‘greatly depleted’ inshore stocks.

The recreational fishery is equivalent to approximately 5% of the commercial catch although this percentage is substantially higher in shallow water adjacent to popular holiday locations. Frusher (1997) estimated the recreational catch off eastern

Tasmania to be equivalent to half of the commercial catch from shallow water regions.

1.3 Current Management

The commercial fishery operates under an individual transferable quota (ITQ) management system that was introduced in 1998 (Anon, 1997). Since implementation of quota management, the Tasmanian catch has been set at 1500 tonnes. The commercial fishing fleet comprises approximately 280 vessels that are licensed to use between 15 and 50 traps in 1999. The Tasmanian fishery has a male and female minimum legal size limit of 110 mm and 105 mm carapace length, respectively. Closed seasons, trap size restrictions and escape gaps in traps are further management arrangements. Details of these are found in Fisheries (Rock Lobster) Rules 1997 (Anon, 1997).

In the Tasmania recreational fishery there were approximately 6153 recreational trap licences and 3465 recreational dive licences issued in the 1996/97 fishing season (Anon, 1997). Recreational trap licences allow the participant to operate one trap under the same regulations of size limits, closed seasons, trap restrictions and escape gaps as for the commercial fishery. In addition there is a daily catch and possession limit (Anon 1997).

1.4 Current assessment techniques

Over the last two decades there has been considerable effort to develop models to assess rock lobster stocks (Walters et al., 1993; Walters et al., 1997; Punt and Kennedy, 1997). The latest 'wave' of models incorporate harvest strategy evaluation procedures whereby the model predicts outcomes (eg. biomass and/or egg production trends) from suggested harvests. Prediction of future trends is based on historic trends. That is, it assumes that what happened in the past will continue to happen in the future. A degree of flexibility to enable future patterns of fishing effort to change is achievable. This comes from the incorporation of 'fleet dynamic' submodels into the assessment models. These submodels are used to 'predict' the movements of fishers from region to region as biomass or other drivers of fishers' behaviour change.

While these models have proved to be extremely useful assessment tools, they need to be constantly validated. As has been seen throughout the history of the rock lobster fishery, rapid changes have occurred. The greatest drivers of change have been technology which has effected fishing patterns directly by enabling fishers to fish more efficiently (GPS, synthetic ropes, trap haulers) or to find and operate in new grounds (trap haulers, echosounders) and indirectly by altering markets (refrigeration, air transportation). In addition to technology, an improved understanding of rock lobster biology and the skill base of fishers' have contributed to their pattern of fishing. The imposition of government regulations also effects the behaviour of fishers. The latest management change (the individual transferable quota system) is expected to have a profound impact on fishers' behaviour. For the first time in the Tasmanian rock lobster fishery, fishers will be increasing their profits by increasing the efficiency of their operations (i.e. improving the profit obtained per kilogram) rather than by improving their total catch.

To provide validation of the assessment models, the Tasmanian Government implemented a 'fisheries independent' sampling project in 1992. A major objective of this project was to obtain regional estimates of exploitation rate and biomass. Two methods, change-in-ratio (CIR: Paulik and Robson, 1969; Pollock and Hoenig, 1998) and index-removal (IR: Eberhardt, 1982; Hoenig and Pollock, 1998) were trialed in northwestern, eastern and southern Tasmania. The sampling strategies and assumptions of these methods are described in Frusher et al (1998). These methods make the explicit assumption that the decline in relative (CIR) or absolute (IR) abundance from a pre-season to a post-season survey is dependent on exploitation rate. Migration, natural mortality and changes in catchability would also affect the decline in the abundance. The CIR method can accommodate these changes providing they effect the harvested and non-harvested components used in determining the ratio equally. To minimise the possibility of differential impacts on the harvested and non-harvested components used in the CIR method, Frusher et al. (1998) used a narrow size class above and below the minimum legal size limit of the fishery. Despite the adoption of the narrow size limits, there are occasions when variable and inconsistent estimates of exploitation rates, using the CIR and IR methods, are obtained. To determine the suitability of the data for estimating exploitation rates, two simple diagnostic tests were developed to validate the assumptions relating to catchability

that are required to be satisfied before using the methods. These diagnostic tests are demonstrated with examples from the south and east coasts in Chapter 2.

The CIR and IR methods proved ineffective in northwestern Tasmania where lobsters undertake a second moult during the fishing season and moulting periods are more protracted. When moulting occurs during the season, the post-season survey for estimating exploitation rates captures lobsters that have been available for exploitation for varying periods of time. Annual exploitation rate, by definition refers to those lobsters captured during the fishing season that were available for harvest from the commencement of the fishing season. Identifying the fraction of lobsters harvested can be achieved by tagging and Chapter 3 describes and trials a method for obtaining fishing mortality from tagging studies.

Although the sampling strategy in northwestern Tasmania was designed for estimating exploitation rates using CIR and IR techniques, the relative standard errors of the fishing mortality estimates derived using the tagging model were low. In contrast the relative standard errors of natural mortality estimates was high and Chapter 4 evaluates different sampling designs for improving the relative standard errors of both fishing and natural mortality estimates and for improved cost efficiency.

In addition to validating the outputs of assessment models, there is a constant need to validate the appropriateness of model inputs. In the Tasmanian rock lobster assessment model, Punt et al. (1997) found regional anomalies in size specific selectivity curves, which they could not explain. Despite these anomalies, the curves were the best that could be estimated from the available data and continue to be used in the model.

Chapter 5 investigates size-specific selectivity from southern and eastern regions of Tasmania and also from a reserve population where larger lobsters are more common. Chapter 6 discusses the implications that the findings on selectivity have for interpretation of changes in the size structure of the fishery.

The final chapter (Chapter 7) summarises the outcomes of these studies and the implications for the current assessment of the Tasmanian rock lobster fishery.

2. Evaluating the Performance of Change-in-Ratio and Index-Removal Estimators of Exploitation Rate in the Southern Rock Lobster Fishery of Tasmania

2.1 Abstract

Concern over the assumptions about catchability has probably inhibited scientists from using the change-in-ratio (CIR) and index-removal (IR) methods for determining exploitation rate in fisheries. Two diagnostic tests based on the difference in size-specific catch rates of sublegal lobsters (IR) and size-specific standardised catch (CIR) were compared to graphical representations to determine cut-off values for when the technique could be used. Cut-off values developed for southern populations of male lobsters were then applied to East Coast populations to see if the proposed criteria provided reasonable results. Analysis of the East Coast population involved using broader size ranges and both sexes. The cut-off values were supported by the East Coast analysis. Application of the diagnostic tests found the assumptions of catchability to be violated for nearly half of the sampling periods. Despite this, exploitation rate estimates were available for 5 of the 6 fishing seasons for males from southern and eastern regions of the fishery and for 4 of the 6 fishing seasons for females on the east coast. The diagnostic tests also indicated where the assumption of a closed population was violated by an earlier than expected moult at the end of the fishing season on the south coast. The CIR technique, with a weaker assumption regarding catchability, was appropriate more often than the IR technique. However, as the data collected can be used for both techniques, it is suggested that both techniques are worthy of greater consideration by fishery biologists.

2.2 Introduction

Biomass and fishing mortality estimates are commonly used as reference points and performance indicators in fishery management. It is thus surprising that the change-in-ratio (CIR)(see review Pollock and Hoenig 1998) and index removal (IR)(see review

Hoenig and Pollock 1998) methods, which provide estimates of exploitation rate, have received only limited attention in the fisheries literature (Dawe et al., 1993, Chen et al., 1997, Frusher et al., 1997, 1998). This is despite various authors suggesting their potential use in fisheries (Chapman 1961, Paulik and Robson 1969, Ricker 1975). While it is uncertain why there has been a reluctance to use these methods, we suggest that concerns over the assumption of constant catchability over time in the IR method, and equal catchability between the two components used in the CIR method, may discourage scientists from testing these methods.

We propose a series of simple diagnostic tests that can be applied to the data to determine its suitability. We test these methods against data obtained from a fisheries independent study of the Tasmanian rock lobster fishery in which pre-season and post-season surveys were conducted from 1992 to 1998.

2.3 The IR and CIR Methods

2.3.1 *Index-removal*

The IR estimator of exploitation rate (U_{IR}) is

$$(1) \quad U_{IR} = \frac{c_1 - c_2}{c_1}$$

where c_1 and c_2 are the catch rates of legal sized animals in the pre-season and post-season surveys, respectively. Hoenig and Pollock (1998) listed the assumptions for the IR method: 1) the population is closed between surveys except for harvest and thus there is no net change to the population through immigration, emigration or recruitment (in crustaceans this is equivalent to moulting), and 2) animals are equally catchable during each survey and among surveys.

In most lobster fisheries there is a mandatory minimum size limit. If the above assumptions hold then lobsters below this limit would be expected to have the same catch rate during both surveys. Comparison of the catch rates of the sub-legal

components over time would indicate whether the IR method could be validly applied to the data. We propose that a table of deviations be constructed based on the survey to survey differences between the catch rates for each of the sublegal size classes. To standardise the catch rates so results can be compared across regions and years, we divided the differences in the catch rates by the pre-season catch rate. The standardized deviation for the i^{th} sublegal size class is

$$(2) \quad d_i = \frac{c_{1i} - c_{2i}}{c_{1i}}$$

where c_{1i} and c_{2i} are the catch rates of animals in the i^{th} size class in surveys 1 and 2 respectively.

In different regions different numbers of undersized size classes may be used to provide adequate sample sizes. We have divided the summed value of the deviations by the number of size classes to account for the number of size classes. This allows one to compare data from different regions and years. The aggregated deviation is

$$(3) \quad \bar{d}_{IR} = \frac{\sum_{i=1}^K d_i}{K}$$

where K is the number of size classes.

To determine if the results from the deviation analysis were providing meaningful and consistent results we compared the deviation analysis to plots of the undersized catch rates for each of the surveys. Lower deviation values would be expected to occur when trends in the plots of sublegal catch rates were similar during each of the surveys. Although subjective, visual observation appeared the best way of validating the deviation analysis.

2.3.2 *Change-in-ratio*

The CIR method is based on changes in the ratio of abundance of two or more components of the population over time (Kelker 1940; Pollock and Hoenig 1998).

Applications involving estimation of population size are found primarily in the wildlife literature, although recently it has been used in fisheries studies by Chen et al. (1997), Dawe et al. (1993), Frusher et al. (1997, 1998). Paulik and Robson (1969) derived a CIR estimator of exploitation rate but there has been limited attention given to this approach. The CIR estimator of exploitation rate (U_{CIR}) of the legal sized component when the sublegal component of the fishery is unharvested is:

$$(4) \quad U_{CIR} = \frac{p_1 - p_2}{p_1(1 - p_2)}$$

where p_i = proportion of legal sized animals in the catch in the i^{th} survey. Survey 1 is undertaken at the start of the period of exploitation and survey 2 at the end of the period of exploitation. We show (Appendix 2.1) that, if the two components of the population have a constant ratio of catchability over time, then the estimate of exploitation rate will be unbiased. That is, for the special case where one component is not harvested, it is not necessary to assume equal catchability of the two components. However, if the ratio of catchabilities varies between surveys then a bias is created in the estimate of exploitation rate. For any change in the ratio of catchabilities over time, the bias is greater the lower the exploitation rate (Appendix 2.1).

To determine the suitability of survey data for analysis with the CIR method we propose that a table of deviations be constructed based on the differences between the standardized length frequency distributions based on numbers caught per size interval. Standardisation is achieved by dividing the catch in each size class by the catch from the sublegal class with the highest catch from the corresponding survey. The standardised number of lobsters caught in the i^{th} sublegal size class is

$$(5) \quad n_i = \frac{l_i}{\max l_i}$$

where l_i is the number of lobster caught in the i^{th} size class. A legal-sized size class can not be used to standardise the data as the legal sized catch is affected by exploitation once the fishing year commences. The standardised deviation for the i^{th} sublegal size class is

$$(6) \quad d_i = \frac{n_{1i} - n_{2i}}{n_{1i}}$$

where n_{1i} and n_{2i} are the standardised number of lobsters caught in the i^{th} size class in surveys 1 and 2 respectively. To account for different size classes being used in the analysis we have divided the sum of the standardised deviations by the number of size classes. The aggregated deviation is

$$(7) \quad \bar{d}_{CIR} = \frac{\sum_{i=1}^K d_i}{K}$$

where K is the number of size classes.

To determine if the results from the deviation analysis were providing meaningful and consistent results we compared the deviation analysis to plots of the sublegal standardised numbers between the surveys. Lower deviation values would be expected to occur when plots of the standardised numbers of sublegal animals caught are similar during each of the surveys. When the patterns of standardised numbers caught versus size for sublegal-sized animals differ between the two surveys, there is evidence that size-specific catchability has changed and the method may produce biased results. Although subjective, visual observation appeared the best way of validating the deviation analysis.

Although the absolute values of residuals, or the squares of residuals, are normally summed to give an indication of goodness of fit of a model, violations in the assumption of constant catchability, inherent in the CIR and IR methods, are

demonstrated by patterns in the differences, rather than the magnitudes of the differences. Thus standardised curves that are either consistently above or below the pre-season curve indicate a catchability problem rather than a curve being inconsistently above and below the pre-season curve which suggests noise associated with, for instance, low sample sizes rather than a catchability violation. However, while summing the differences between two curves may lead to detection of some problems, it is also possible it will obscure some problems. For example, observation of three successive size classes with negative differences followed by three successive size classes with positive differences might be suggestive of a trend over size and thus indicate a problem; but, the sum of these differences might be close to zero and thus not reflect the problem. For this reason, the proposed criterion for detecting problems should only be used in conjunction with a visual examination of the two curves.

When the assumptions for both the CIR and IR methods are met, the estimates should be identical except for sampling errors. It is interesting to consider what happens when natural mortality and recruitment occur during the fishing season. The IR method is based on total removals (including recruitment, which can be considered a negative removal) and thus estimates the proportional change in the legal-sized population due to fishing and natural mortality and recruitment. In contrast, the CIR method is based on the change in legal-sized animals relative to the change in sublegal animals. For example, if natural mortality is equivalent for both components used in the estimation of exploitation rates, and there is no recruitment, then the removal (change) only includes fishing mortality. Hence, the CIR method would estimate the proportional change in population due to fishing. (Effects of recruitment are more complicated because it depends on the relative recruitment to each of the two groups.) In determining exploitation rates in the Tasmanian rock lobster fishery, Frusher et al (1998) used lobsters just below and just above the minimum legal size. Natural mortality can be considered equal for these two components because their sizes are so similar. Recruitment (moulting) in southern regions of the Tasmanian rock lobster fishery normally occurs during the closed season although, as will be demonstrated, it can occur just prior to the closure of the fishing season. Tagging data indicate migrations of lobsters to be negligible (Pearn, 1994). Thus the population is essentially closed and the assumption of equal catchability is the most likely assumption to cause any bias in exploitation rate estimates. As the CIR method is

unaffected by change in catchability between surveys (provided the ratio of catchability remains constant over time), the weaker assumption of the CIR method is more likely to be met in practice than the IR assumption.

2.4 Application to Southern Rock Lobster in Tasmania

We now apply these tests to fishery independent catch sampling surveys undertaken at the start, middle and end of the rock lobster fishing season in southern and eastern Tasmania.

2.4.1.1 Southern Tasmania

In southern Tasmania the fishery is primarily based on males as few females grow to legal size (Frusher, 1997). The exploitation rates obtained by both the IR and CIR methods are similar for several fishing years (e.g., first and second halves of the 92/93 fishing season and first half of the 95/96 season) but vary substantially for others (e.g., second half of most fishing seasons)(Figure 2.1). In all six years, the IR estimate was higher than the CIR estimate for the second half of the fishing season. In contrast, for the first half of the fishing year the IR estimate was higher in three years and the CIR estimate was higher in three years.

For the IR method, comparison of the size-specific catch rates show that there was only one occasion when the catch rate of sublegal lobsters was equivalent between surveys (start and middle of 1995/96 fishing year)(Figure 2.2). This is supported by the low deviation value of 0.091 for this pair of surveys (Table 2.1). The next lowest value is -0.161, which was obtained from the first half of the 1993/1994 fishing season. There is an obvious deviation between the plots of the catch rates for this value. We therefore suggest that a value of 0.100 may be an appropriate benchmark for judging when the catchability assumption is tenable.

In comparison, for the CIR method, the plots of standardised numbers caught per size class show that there was a similarity between the sublegal portions of the catch on more occasions (e.g., start and middle for all except the 1993/94, 1994/1995 and 1995/1996 fishing years and start and end for 1992/93, 1993/94 and 1996/97 fishing years) (Figure 2.3). Although the sublegal standardised size frequency is similar for

the start and end of the 1997/98 fishing year, there is an increase in the number of legal sized lobsters caught at the end of year survey compared to the middle of year survey. This is evident in the increased standardised legal-sized component of the graphs and in the negative residuals of the legal-sized component between the middle and end of year survey. This implies that either the sublegal size classes were underrepresented in the catch (e.g., because of moulting affecting availability to the gear) or there had been recruitment to the fishery (e.g., lobsters having completed a moult from undersized to legal sized and now being available to the fishing gear). The latter appears to have been the case as the annual male moult occurred early near the end of the 1997/98 fishing year. Both fishers and processors reported soft shelled lobsters indicating that moulting had recently occurred. It also occurred in the 1994/95 fishing year and legal sized catch also increased after the middle of year survey (Figure 2.3).

The table of deviations shows the highest value for the supported plots to be 0.050 (Table 2.2). The next highest value (0.057) is for the first half of the 1994/1995 fishing season where there is an observed difference between the graphs. The suggested deviation value to support use of the CIR method is considered to be 0.050 or less. A value of -0.015 was obtained for the start to end of the 1995/1996 fishing season. This was not supported by the trend in the graphs for the start and end of fishing year surveys in 1995/96. However, the differences are both positive and negative and cancel each other. The different signs in the column for the undersized portion of the catch in Table 2.2 are more likely to indicate sampling variation than a bias due to catchability. This sampling variation would account for the broad 95% confidence intervals found around the CIR exploitation rate estimate (Figure 2.1) obtained from bootstrapping the data (Frusher et al., 1998).

A low value of 0.010 was also obtained for the start and end surveys for the 1997/1998 fishing season. Analysis of the deviations in the legal sized component of the catch shows that the values are negative. Therefore an increase in the number of legal sized lobsters has occurred between the middle and end of season surveys. This highlights the importance of checking the legal sized deviations for any systematic pattern that may indicate a violation of the assumptions. Although the information can not be used to estimate exploitation rate it does provide additional information about

the population of lobsters. In this case, the population wasn't closed and recruitment (moulting to legal size) was occurring. Thus information is gained regarding the timing of the moult or, as is the case in the Tasmanian fishery, that an earlier than expected moult was occurring.

2.4.1.2 *East Coast*

We now test the cut off values of the deviations established for the south coast on the exploitation rates data for the east coast. Advantages of using the east coast data are that they are from a different region, both males and females are harvested, exploitation rates are obtained for each sex separately, and the number of sublegal size classes used to estimate exploitation rates are greater than for the south coast region.

For the IR method, only four of the aggregated deviations using the catch rates for males are below 0.100 (Table 2.3). These seasons are also supported by the graphs (Figure 2.4). Thus the start to middle of the 1996/1997 and 1997/1998 fishing seasons and the start to end of the 1992/1993 and 1996/1997 fishing seasons are suitable for estimating exploitation rate of males lobsters on the east coast using the IR method. None of the catch rate deviations were below 0.100 for female lobsters (Table 2.4) and graphs of the catch rates also indicated substantial changes in the catch rate of sublegal lobsters between the start and middle surveys (Figure 2.5) (Female lobster moult after the middle of year surveys and the fishing season closes. Hence there are no estimates of exploitation for the second half of the season). The IR method appears to be inappropriate for estimating exploitation rate for female lobsters on the east coast of Tasmania.

To determine the suitability of the survey data to estimate exploitation rate using the CIR method we find 5 sampling periods for males (Table 2.5) and four sampling periods for females (Table 2.6) to be below the 0.050 criterion established from the south coast analysis. These values are also supported by the respective graphs (Figure 2.6 and 2.7). For males, the start and end surveys for the 1994/1995 and 1995/1996 surveys had values just above the 0.050 cut off value of 0.063 and 0.067 respectively. These graphs indicate an observable difference between the undersized distributions and thus support our cut off value.

There are three occasions when both the CIR and IR method passed their respective diagnostic tests for the same period. Exploitation rates estimated by both methods are very close for each of these periods (Figure 2.8).

Although the analysis supports the CIR estimates of exploitation rate off the East Coast for the start to middle and start to end of the 1996/1997 fishing year, the exploitation rates appear erroneous as they are identical. A small change in exploitation rates would be expected as approximately 20% of the annual harvest is undertaken during the second part of the fishing season. However the east coast region, where the exploitation rate estimates apply, accounts for around 5 % of the total TACC (Frusher and Gardner, 1999) and thus 20% of the catch in this region does not represent a significant number of lobsters. It is therefore possible that the effort expended by the commercial fleet between March and August may not have occurred in the survey areas. Unfortunately the detail available in the commercial fishing logbooks is not sufficient to determine the amount of effort directed at the specific survey sites.

Although only 21 of the possible 60 estimates of exploitation rate passed the diagnostic tests, there was only one of the six fishing seasons on the south (1994/1995) and east coast (1993/1994) that exploitation rate estimates were not available for at least part of the fishing season. Exploitation rates for females were available for 4 of the six fishing seasons.

Although our empirical approach is somewhat arbitrary, it does provide a basis for discerning whether the assumptions of catchability are being violated, although further attention should be devoted to developing improved diagnostic procedures.

We conclude that with the application of a couple of simple diagnostic tests it is possible to determine the suitability of size structure data for determination of exploitation rates using the CIR and IR methods. Given the importance of fishing mortality and biomass estimates as fishery reference points and performance indicators, it is considered prudent to obtain estimates from as many different sets of data as possible. It is not uncommon for exploitation rate estimates to have broad confidence limits and thus robustness of the point estimates is then derived from the

similarity of the point estimates obtained from different data sources using different methods.

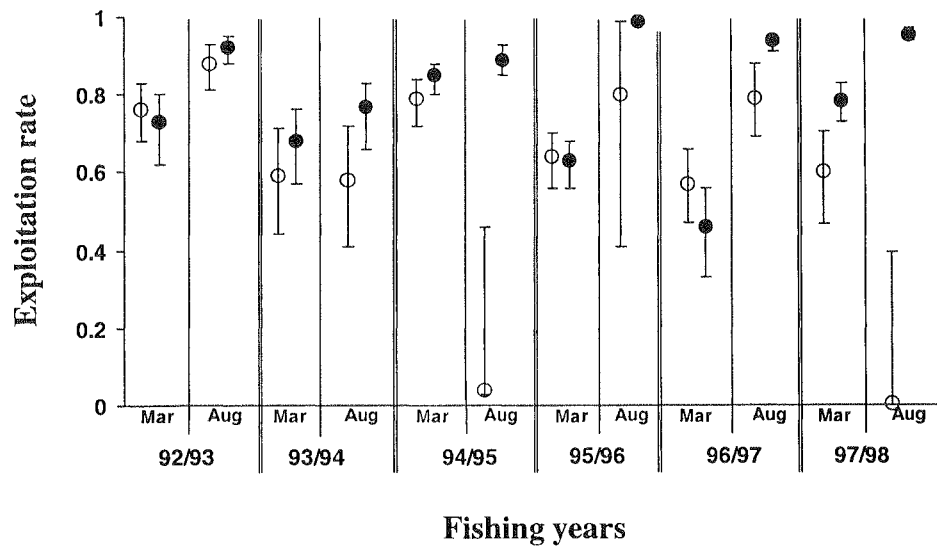


Figure 2.1. Exploitation rates estimated for the first (Mar) and second (Aug) halves of the fishing year by the CIR (open circle) and IR (closed circle) methods in southern Tasmania for the 1992/1993 to 1997/1998 fishing years. Error bars are 95% confidence limits determined by bootstrapping the data (Frusher et al. 1998).

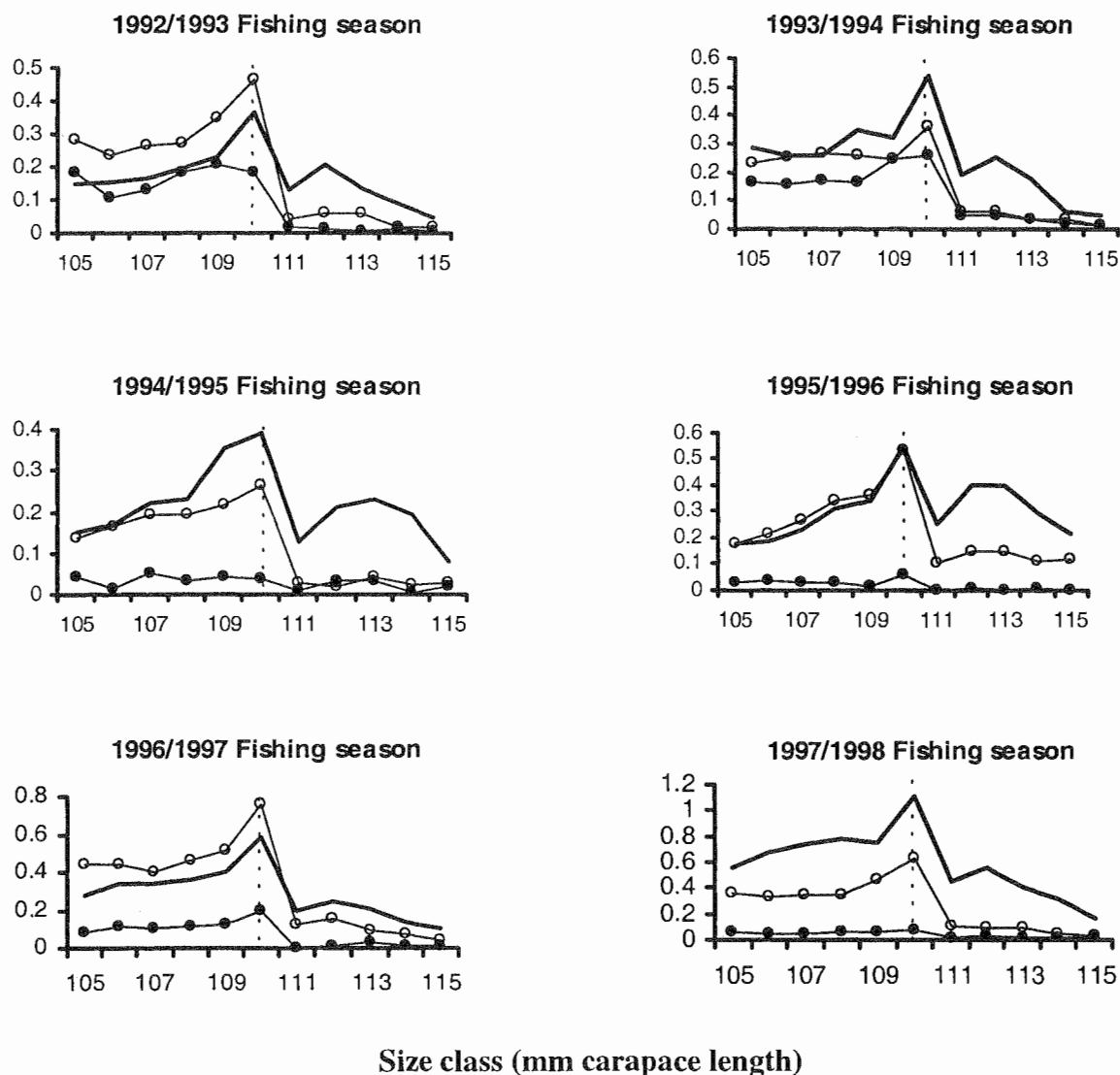


Figure 2.2. Comparison of catch rates (numbers/trap lift) of male lobsters caught in southern Tasmania during surveys undertaken at the start (thick line, no symbol), middle (open circle) and end (closed circle) of the fishing years from 1992/1993 to 1997/1998. Vertical dashed lines show the minimum legal size limit.

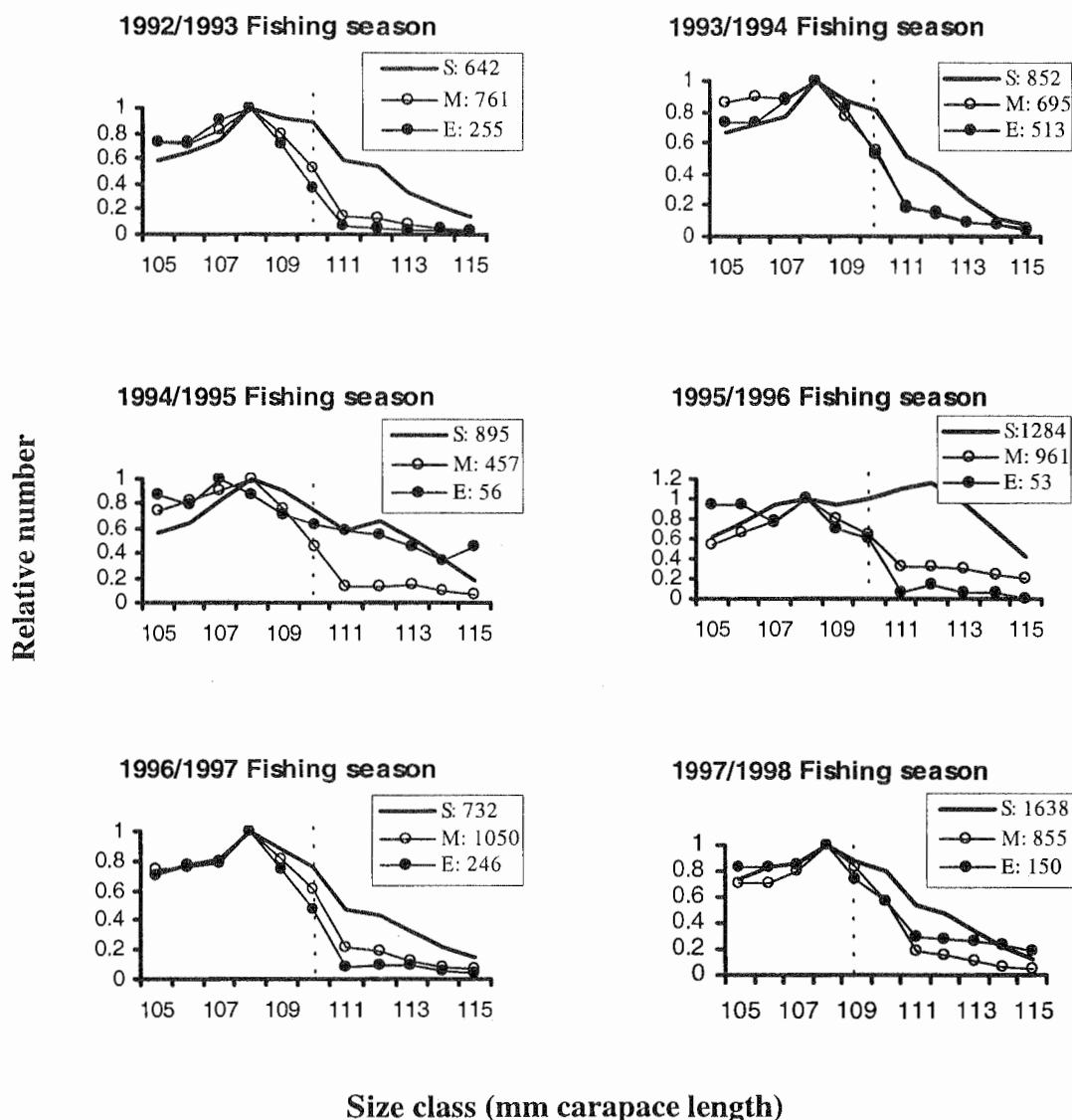


Figure 2.3. Standardised size frequency distribution of male rock lobsters caught in southern Tasmania from 105mmCL to 115mmCL for surveys at the start (S), middle (M) and end (E) of the fishing year from 1992/93 to 1997/98. Relative number is obtained by dividing the catch in each sized class by the maximum catch in the sublegal size class for each survey. Vertical dashed lines show the minimum legal size limit. Numbers in the legend refer to total number of lobsters caught in the survey for the 105mmCL to 115mmCL size classes.

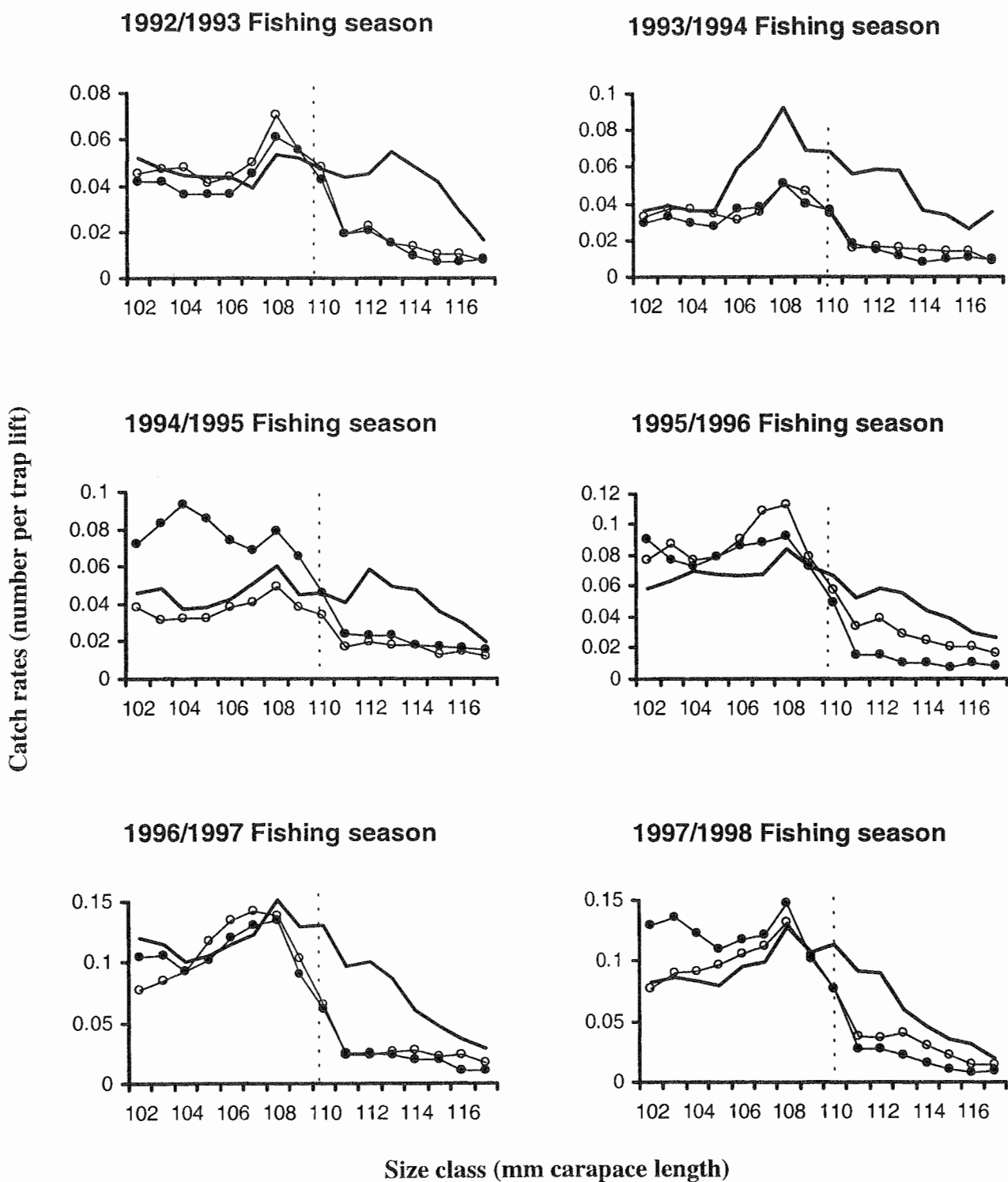


Figure 2.4. Comparison of catch rates of male lobsters caught on the east coast during surveys conducted at the start (no symbol), middle (open circles) and end (closed circles) of the 1992/1993 to 1997/1998 fishing seasons. Vertical lines indicate the minimum legal size.

Catch rates (number per trap lift)

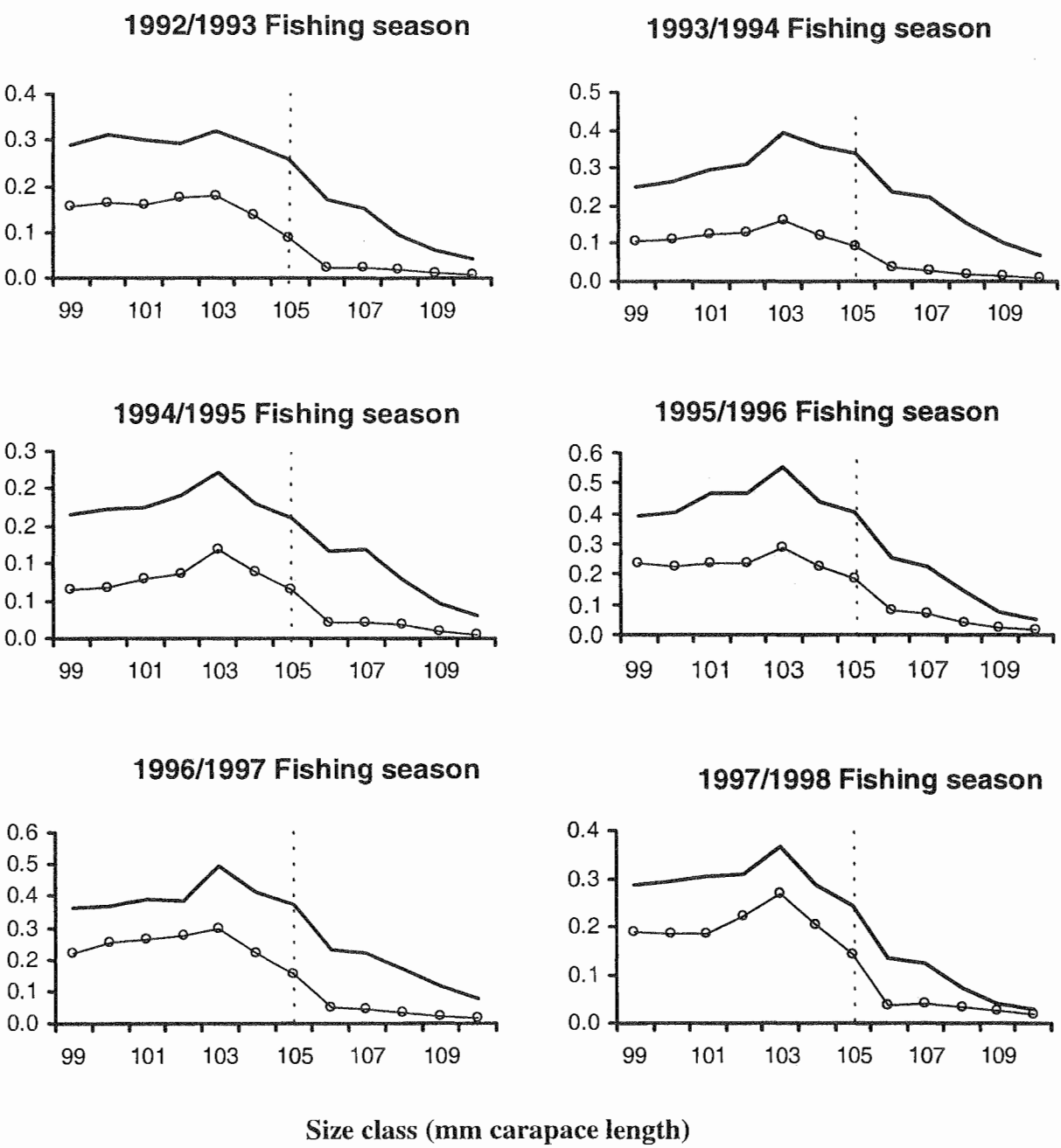


Figure 2.5. Comparison of catch rates (numbers/trap lift) of female lobsters caught on the east coast during surveys conducted at the start (no symbol) and middle (open circles) of the 1992/1993 to 1997/1998 fishing seasons. Vertical lines indicate the minimum legal size.

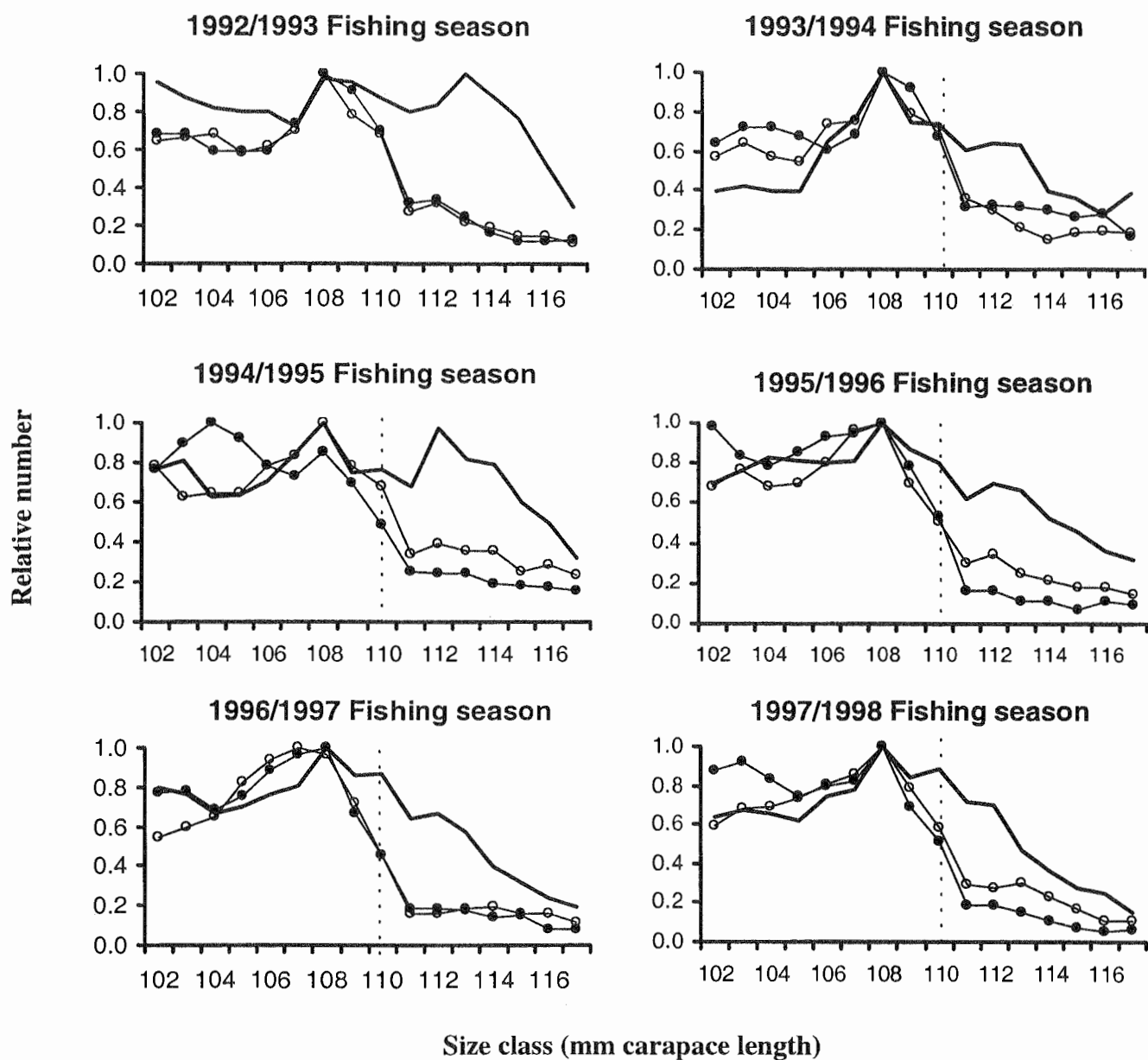


Figure 2.6. Comparison of relative number of male lobsters caught on the east coast during surveys conducted at the start (no symbol), middle (open circles) and end (closed circles) of the 1992/1993 to 1997/1998 fishing seasons. Relative number is obtained by dividing the catch in each sized class by the maximum catch in the sublegal size class for each survey. Vertical lines indicate the minimum legal size.

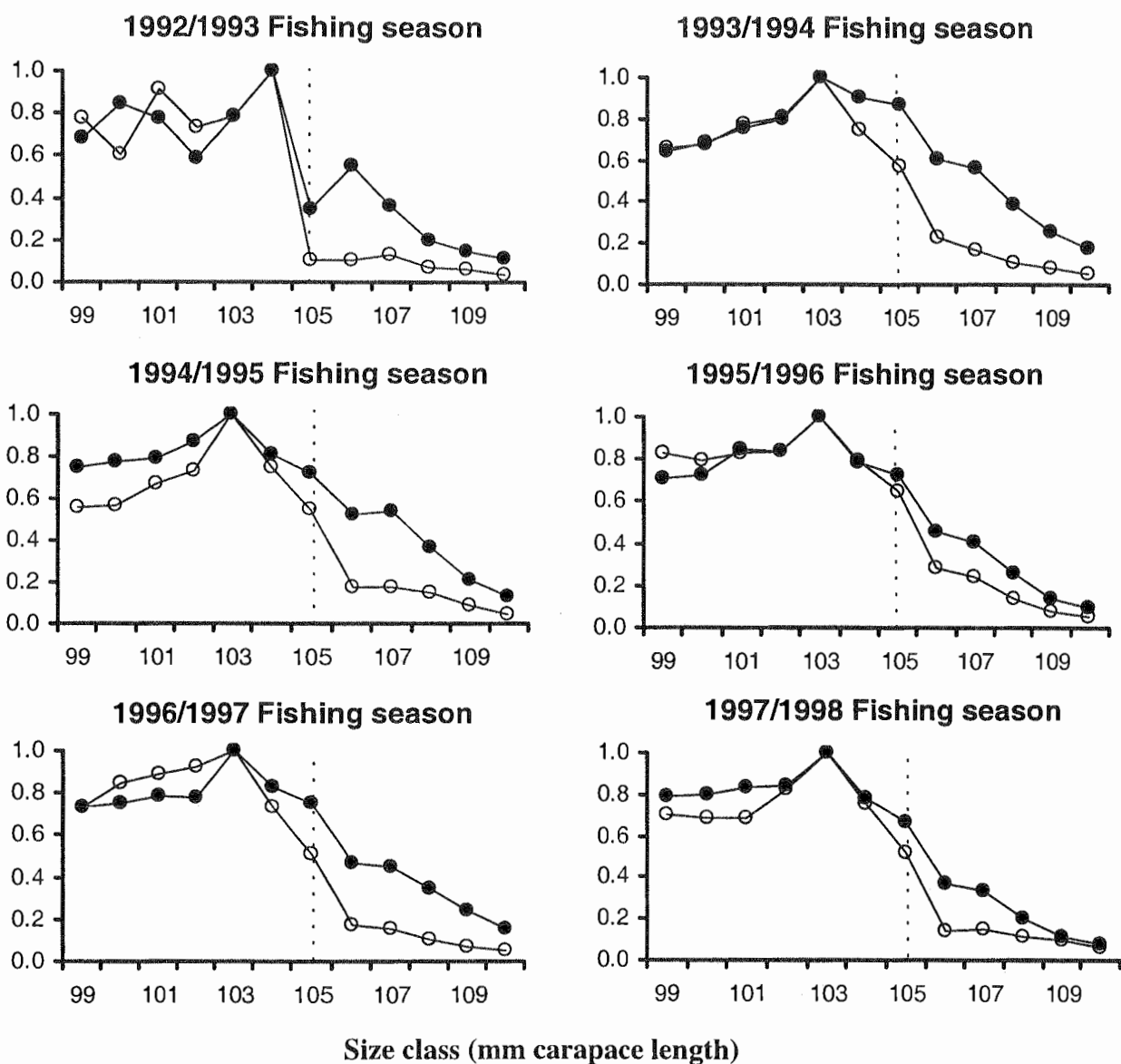


Figure 2.7. Comparison of the relative number of female lobsters caught on the east coast during surveys conducted at the start (no symbol) and middle (open circles) of the 1992/1993 to 1997/1998 fishing seasons. Relative number is obtained by dividing the catch in each sized class by the maximum catch in the sublegal size class for each survey. Vertical lines indicate the minimum legal size.

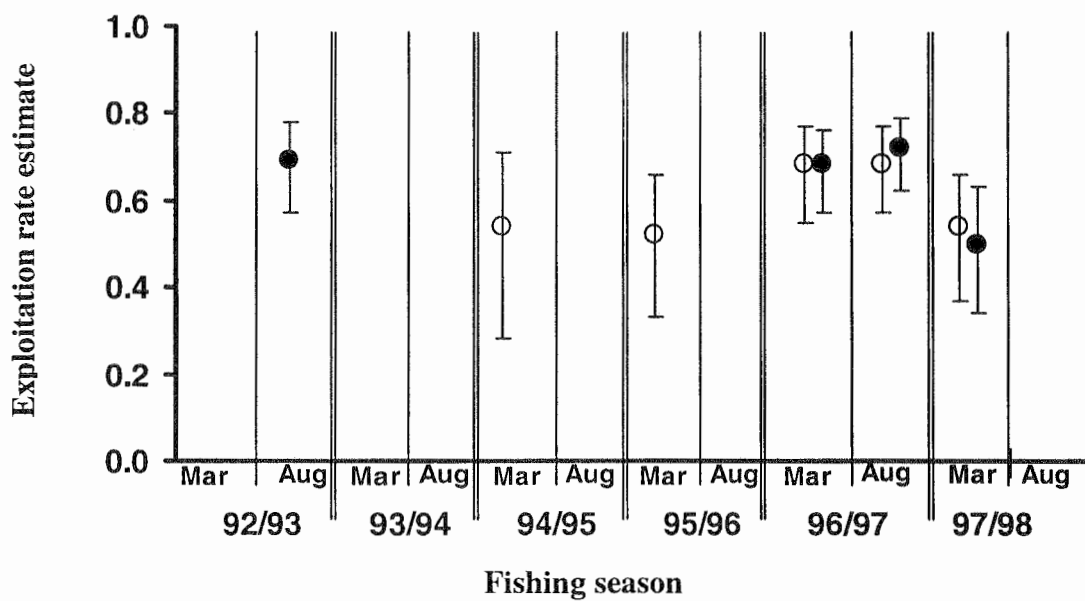


Figure 2.8. Exploitation rate estimates using the change-in-ratio (open circle) and index-removal (closed circles) methods for the east coast when the diagnostic tests were satisfied.

Table 2.1. IR diagnostics: Deviations are differences in the catch rates of lobsters caught in survey 1 and survey 2 standardised by the catch rate in survey 1

Size class	92/3S-M	92/3S-E	92/3M-E	93/4S-M	93/4S-E	93/4M-E	94/5S-M	94/5S-E	94/5M-E	95/6S-M	95/6S-E	95/6M-E	96/7S-M	96/7S-E	96/7M-E	97/8S-M	97/8S-E	97/8M-E
105	0.531	-0.317	-0.554	-0.021	-0.402	-0.389	-0.038	-0.909	-0.905	0.154	-0.811	-0.836	0.322	-0.655	-0.739	-0.516	-0.926	-0.847
106	0.646	-0.218	-0.525	0.035	-0.341	-0.364	-0.118	-0.765	-0.734	0.175	-0.862	-0.883	0.168	-0.693	-0.737	-0.534	-0.941	-0.874
107	0.384	-0.074	-0.331	-0.254	-0.520	-0.356	-0.147	-0.864	-0.840	0.108	-0.898	-0.908	0.273	-0.694	-0.760	-0.565	-0.923	-0.824
108	0.518	-0.114	-0.417	-0.236	-0.245	-0.011	-0.382	-0.883	-0.810	0.050	-0.949	-0.952	0.282	-0.679	-0.749	-0.379	-0.927	-0.882
109	0.273	-0.503	-0.609	-0.328	-0.519	-0.284	-0.324	-0.907	-0.862	-0.032	-0.898	-0.895	0.298	-0.669	-0.745	-0.438	-0.937	-0.887
Sum of undersized	2.352	-1.227	-2.436	-0.805	-2.027	-1.404	-1.009	-4.327	-4.151	0.456	-4.418	-4.472	1.343	-3.390	-3.730	-2.432	-4.654	-4.313
Sum of undersized/number of undersized size classes	0.470	-0.245	-0.487	-0.161	-0.405	-0.281	-0.202	-0.865	-0.830	0.091	-0.884	-0.894	0.269	-0.678	-0.746	-0.486	-0.931	-0.863

Table 2.2. Difference between standardise catches of male lobsters caught on the south coast from 105mmCL to 115mmCL between the start and middle (SM), start and end (SE) and middle and end (ME) surveys for the fishing years from 1992/1993 to 1997/1998. Bold figures in the legal-sized portion of the catch (≥ 110 mmCL) highlight negative differences.

Size class	1992/1993			1993/1994			1994/1995			1995/1996			1996/1997			1997/1998		
	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME
105	-0.14	-0.14	0.00	-0.20	-0.07	0.13	-0.17	-0.31	-0.14	0.09	-0.31	-0.40	-0.03	0.01	0.05	0.03	-0.08	-0.11
106	-0.07	-0.08	-0.02	-0.18	-0.02	0.16	-0.18	-0.15	0.03	0.10	-0.17	-0.27	0.02	0.01	-0.01	0.12	0.00	-0.12
107	-0.07	-0.16	-0.09	-0.12	-0.10	0.02	-0.07	-0.17	-0.10	0.15	0.17	0.02	0.03	0.02	0.00	0.05	0.00	-0.05
108	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0
109	0.13	0.20	0.07	0.10	0.05	-0.05	0.14	0.19	0.05	0.13	0.24	0.10	0.07	0.13	0.06	0.04	0.13	0.09
Sum of undersized	-0.15	-0.19	-0.03	-0.39	-0.14	0.26	-0.29	-0.33	-0.04	0.46	-0.08	-0.54	0.08	0.17	0.09	0.25	0.05	-0.20
Mean	-0.031	-0.037	-0.007	-0.079	-0.027	0.052	-0.057	-0.066	-0.008	0.093	-0.015	-0.108	0.017	0.035	0.018	0.050	0.010	-0.058
110	0.36	0.51	0.15	0.25	0.29	0.03	0.29	0.12	-0.17	0.36	0.39	0.03	0.16	0.29	0.13	0.23	0.23	-0.004
111	0.45	0.53	0.08	0.33	0.32	-0.01	0.46	0.00	-0.45	0.78	1.04	0.25	0.26	0.40	0.14	0.35	0.24	-0.111
112	0.42	0.49	0.08	0.26	0.26	0.01	0.53	0.11	-0.42	0.83	1.03	0.19	0.25	0.33	0.09	0.34	0.20	-0.138
113	0.26	0.30	0.05	0.15	0.15	0.00	0.38	0.06	-0.32	0.66	0.90	0.24	0.21	0.23	0.02	0.23	0.08	-0.155
114	0.17	0.18	0.01	0.05	0.04	0.00	0.26	0.02	-0.24	0.43	0.61	0.18	0.14	0.16	0.02	0.15	-0.02	-0.168
115	0.10	0.11	0.01	0.04	0.02	-0.01	0.11	-0.28	-0.39	0.22	0.42	0.20	0.09	0.11	0.02	0.08	-0.07	-0.143

Table 2.3. East coast males. Deviations are differences in the catch rates of lobsters caught in survey 1 and survey 2 standardised by the catch rate in survey 1.

Size class	92/3S-M	92/3S-E	92/3M-E	93/4S-M	93/4S-E	93/4M-E	94/5S-M	94/5S-E	94/5M-E	95/6S-M	95/6S-E	95/6M-E	96/7S-M	96/7S-E	96/7M-E	97/8S-M	97/8S-E	97/8M-E
102.000	0.432	0.190	-0.426	0.081	-0.066	-0.160	0.592	0.113	-1.171	-0.475	-0.583	-0.073	0.394	0.039	-0.586	0.130	-0.394	-0.603
103.000	-0.571	-0.148	0.269	0.166	0.214	0.058	-0.003	-1.261	-1.255	-0.420	-0.225	0.137	0.261	0.229	-0.043	-0.027	-0.599	-0.557
104.000	-0.121	0.256	0.337	0.238	-0.072	-0.407	0.311	-1.618	-2.799	-0.219	0.115	0.274	0.072	-0.054	-0.136	-0.297	-0.771	-0.365
105.000	0.305	0.367	0.090	0.190	-0.357	-0.675	0.141	-1.565	-1.985	0.269	-0.024	-0.400	-0.145	0.029	0.151	0.030	-0.098	-0.132
106.000	-0.145	-0.283	-0.120	0.271	0.372	0.139	0.060	-0.559	-0.658	-0.621	-0.659	-0.023	-0.208	0.111	0.264	-0.445	-0.320	0.086
107.000	-0.298	0.256	0.427	0.469	0.757	0.543	0.093	-0.283	-0.414	-0.803	-0.225	0.321	-0.121	-0.250	-0.115	-0.018	-0.286	-0.263
108.000	-0.360	-0.493	-0.098	0.604	0.204	-1.010	0.373	-0.247	-0.990	-0.416	-0.064	0.248	-0.171	-0.093	0.067	-0.046	-0.085	-0.037
109.000	-0.331	-0.189	0.107	0.347	0.299	-0.074	0.072	-0.404	-0.512	-0.021	-0.058	-0.037	0.324	0.398	0.109	-0.029	-0.098	-0.067
Sum of deviations	-1.090	-0.043	0.586	2.366	1.352	-1.586	1.639	-5.823	-9.785	-2.705	-1.722	0.447	0.406	0.409	-0.288	-0.701	-2.650	-1.938
Mean deviation	-0.136	-0.005	0.073	0.296	0.169	-0.198	0.205	-0.728	-1.223	-0.338	-0.215	0.056	0.051	0.051	-0.036	-0.088	-0.331	-0.242

Table 2.4. East coast females. Deviations are differences in the catch rates of lobsters caught in survey 1 and survey 2 standardised by the catch rate in survey 1.

	92/3-SM	93/4-SM	94/5-SM	95/6-SM	96/7-SM	97/8-SM
99.000	0.397	0.473	0.615	0.400	0.355	0.227
100.000	0.626	0.622	0.547	0.434	0.406	0.548
101.000	0.379	0.647	0.661	0.474	0.176	0.326
102.000	0.346	0.470	0.453	0.552	0.351	0.328
103.000	0.470	0.629	0.548	0.413	0.325	0.189
104.000	0.472	0.629	0.414	0.474	0.476	0.266
Sum of deviations	2.689	3.470	3.238	2.748	2.089	1.884
Mean of deviations	0.448	0.578	0.540	0.458	0.348	0.314

Table 2.5. Difference between standardise catches of male lobsters caught on the east coast from 105mmCL to 115mmCL between the start and middle (SM), start and end (SE) and middle and end (ME) surveys for the fishing years from 1992/1993 to 1997/1998. Bold figures in the legal-sized portion of the catch (≥ 110 mmCL) highlight negative differences.

Size class	1992/1993			1993/1994			1994/1995			1995/1996			1996/1997			1997/1998		
	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME	SM	SE	ME
102	0.310	0.271	-0.038	-0.183	-0.248	-0.066	-0.016	-0.004	0.012	0.019	-0.279	-0.298	0.258	0.024	-0.234	0.052	-0.234	-0.286
103	0.202	0.188	-0.014	-0.221	-0.303	-0.082	0.178	-0.087	-0.266	-0.010	-0.075	-0.065	0.165	-0.021	-0.187	-0.010	-0.243	-0.233
104	0.143	0.227	0.084	-0.183	-0.330	-0.148	-0.019	-0.375	-0.356	0.149	0.046	-0.102	0.022	-0.017	-0.039	-0.036	-0.179	-0.142
105	0.214	0.203	-0.011	-0.150	-0.281	-0.131	-0.005	-0.281	-0.276	0.114	-0.046	-0.159	-0.120	-0.052	0.068	-0.113	-0.120	-0.007
106	0.179	0.203	0.025	-0.092	0.039	0.131	-0.071	-0.077	-0.006	0.000	-0.128	-0.128	-0.174	-0.122	0.052	-0.059	-0.052	0.007
107	0.012	-0.025	-0.037	0.019	0.084	0.066	0.017	0.115	0.098	-0.153	-0.138	0.015	-0.190	-0.159	0.031	-0.077	-0.047	0.029
108	-0.024	-0.024	0.000	0.000	0.000	0.000	0.000	0.152	0.152	0.000	0.000	0.000	0.034	0.000	-0.034	0.000	0.000	0.000
109	0.167	0.039	-0.127	-0.041	-0.173	-0.131	-0.030	0.054	0.083	0.174	0.086	-0.087	0.137	0.192	0.054	0.045	0.144	0.099
Sum of undersized	1.202	1.084	-0.119	-0.852	-1.213	-0.361	0.054	-0.504	-0.558	0.291	-0.533	-0.825	0.132	-0.156	-0.289	-0.198	-0.731	-0.533
Mean	0.150	0.135	-0.015	-0.107	-0.152	-0.045	0.007	-0.063	-0.070	0.036	-0.067	-0.103	0.017	-0.020	-0.036	-0.025	-0.091	-0.067

Table 2.6. Difference between standardise catches of female lobsters caught on the east coast from 105mmCL to 115mmCL between the start and middle (SM), start and end (SE) and middle and end (ME) surveys for the fishing years from 1992/1993 to 1997/1998. Bold figures in the legal-sized portion of the catch (≥ 110 mmCL) highlight negative differences.

Size class	1992/1993	1993/1994	1994/1995	1995/1996	1996/1997	1997/1998
99	-0.097	-0.022	0.187	-0.120	0.001	0.088
100	0.247	0.003	0.210	-0.067	-0.099	0.112
101	-0.138	-0.019	0.126	0.014	-0.104	0.152
102	-0.141	-0.011	0.137	0.004	-0.140	0.021
103	-0.004	0.000	0.000	0.000	0.000	0.000
104	0.000	0.157	0.063	-0.001	0.096	0.027
Sum of undersized	-0.132	0.108	0.723	-0.170	-0.247	0.399
Mean	-0.022	0.018	0.120	-0.028	-0.041	0.066

Appendix 2.1. Effect of differential catchability among classes and over time on CIR estimates of exploitation rate

The CIR method for estimating population size is based on the assumption that the two components of the population have the same catchability in any survey. However, when only one component of the population is exploited, the method provides unbiased estimates of the size of the exploited component provided the ratio of catchability of the two components stays constant from survey to survey (Seber 1985). This is the first time that a similar analysis of the CIR estimator of exploitation rate has been reported.

Here we investigate the CIR estimator of exploitation rate to see the effect of unequal catchability of the two components. The case where only X-type animals are harvested is considered. Initially, we assume that the ratio of catchability remains constant over time. We then consider the case where the ratio of catchability changes over time.

Suppose the population is composed of X animals of type **X** and Y animals of type **Y**, and let Π_i denote the proportion of animals that is of type **X** in survey i for $i = 1, 2$. If the population is closed except for the removal of **X** type animals, then Π_2 is $(X - R_x)/(Y + X - R_x)$ where R_x is the number of **X** type animals removed. Data from pre- and post-season surveys are used to obtain estimates, p_i of the Π_i . We concern ourselves here with the case where the **X** and **Y** type animals have different catchability such that

$$E(p_i) = X_i/(X_i + \delta Y_i)$$

where $E(.)$ denotes expected value and δ is a parameter expressing the degree to which **X** and **Y** type animals differ in catchability.

The CIR estimator of exploitation rate U_x of commercial-sized (type **X**) animals is

$$(1) \quad \hat{U}_x = \frac{p_1 - p_2}{p_1(1 - p_2)}$$

where the $\hat{}$ denotes an estimate. Substituting expected values for the p_i into (1) gives the (asymptotic) expected value of the estimator:

$$\begin{aligned}
 (2) \quad E(\hat{U}_x) &\approx \frac{\frac{X_1}{X_1 + \delta Y_1} - \frac{X_1 - R_x}{X_1 - R_x + \delta Y_1}}{\frac{X_1}{X_1 + \delta Y_1} \left(1 - \frac{X_1 - R_x}{X_1 - R_x + \delta Y_1} \right)} \\
 &= \frac{X_1^2 - R_x X_1 + X_1 \delta Y_1 - X_1^2 + R_x X_1 - X_1 \delta Y_1 + R_x \delta Y_1}{(X_1 + \delta Y_1)(X_1 - R_x + \delta Y_1)} \cdot \frac{(X_1 + \delta Y_1)(X_1 - R_x + \delta Y_1)}{X_1 \delta Y_1} \\
 &= \frac{R_x \delta Y_1}{X_1 \delta Y_1} = \frac{R_x}{X_1} = U_x
 \end{aligned}$$

Thus catchability can be different for the **X** and **Y** components provided the ratio of catchabilities doesn't change between surveys.

To test the effect of a change in relative catchability of the two components between surveys we replace the catchability parameter δ with a survey specific value, δ_i , for $i = 1, 2$. Then (2) becomes

$$\begin{aligned}
 E(\hat{U}_x) &\approx \frac{\frac{X_1}{X_1 + \delta_1 Y_1} - \frac{X_1 - R_x}{X_1 - R_x + \delta_2 Y_1}}{\frac{X_1}{X_1 + \delta_1 Y_1} \left(1 - \frac{X_1 - R_x}{X_1 - R_x + \delta_2 Y_1} \right)} \\
 &= \frac{X_1^2 - R_x X_1 + X_1 \delta_2 Y_1 - X_1^2 + R_x X_1 - X_1 \delta_1 Y_1 + R_x \delta_1 Y_1}{(X_1 + \delta_1 Y_1)(X_1 - R_x + \delta_2 Y_1)} \cdot \frac{(X_1 + \delta_1 Y_1)(X_1 - R_x + \delta_2 Y_1)}{X_1 \delta_2 Y_1}
 \end{aligned}$$

$$= \frac{X_1 \delta_2 Y_1 - X_1 \delta_1 Y_1 + R_x \delta_1 Y_1}{X_1 \delta_2 Y_1}$$

$$= 1 - \frac{\delta_1}{\delta_2} + \frac{\delta_1 R_x}{\delta_2 X_1} \quad = 1 - \frac{\delta_1}{\delta_2} + \frac{\delta_1}{\delta_2} U_x$$

Thus the expected value of the exploitation rate obtained by the CIR method when the ratio of selectivity changes between surveys is a linear function of the true exploitation rate. When the exploitation rate is zero the expected value of the estimator is $1 - \delta_1 / \delta_2$. As the exploitation rate approaches 1.0 the expected value of the estimator also approaches 1.0 (Figure 2.9). Positive changes in catchability, as shown in Figure 2.9 can lead to negative changes in exploitation when very low exploitation rates exist. This is caused by the increase in the exploitable component which is greater than the number of animals removed. Such a change could occur if catchability was low during the initial survey (say due to mating, moulting, water temperature) and increased during the second survey.

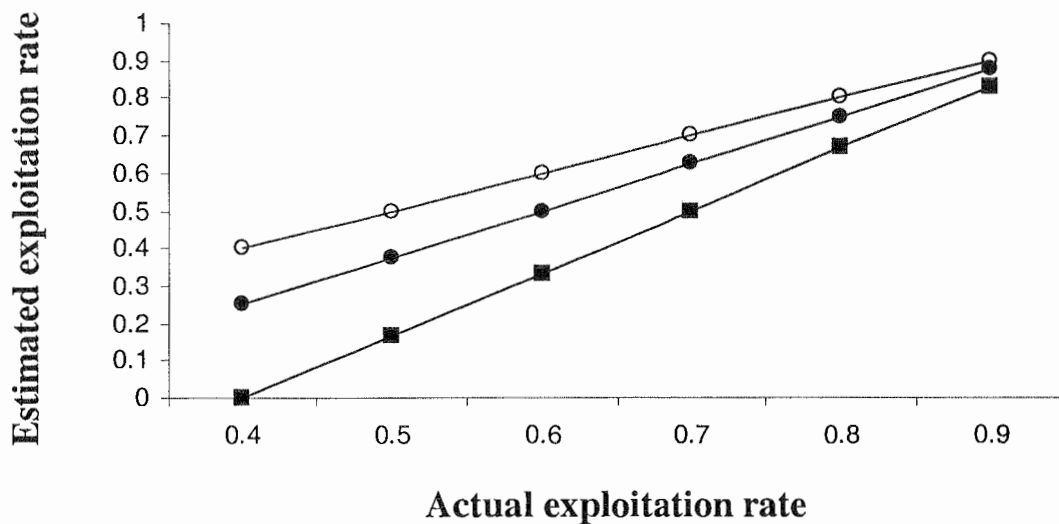


Figure 2.9 Effect on exploitation rate estimates of a change in relative catchability (δ_1 / δ_2) between survey 1 and survey 2. Open circles represent no change in relative catchability ($\delta_1 / \delta_2 = 1$) and the closed symbols represent a change in relative catchability of 1.25 (circles) and 1.67 (squares). The lower the exploitation rate the greater the bias in estimated exploitation rate.

3. *Estimating natural and fishing mortality and tag reporting rate of rock lobster from a multi-year tagging model.*

(This chapter has been accepted for publication in the Canadian Journal of Fisheries and Aquatic Science)

3.1 Abstract

Fishing and natural mortality rates and tag reporting rate for rock lobsters (*Jasus edwardsii*) in northwest Tasmania, Australia, were estimated using multi-year tagging models. These estimates are necessary for assessment of the resource. Several models were examined that had either two or three tagging events each year, and either combined sexes or kept sexes separate. The model that best described the dynamics of the fishery utilized three tagging events within a year. The year was divided into discrete periods and, within each year, fishing effort and duration of period were used to apportion fishing and natural mortalities, respectively, to the periods. The separation of fishing mortalities by sex was not found to improve the models. Although high ($1.0 - 1.2 \text{ yr}^{-1}$), the instantaneous fishing mortality estimates for years where there were sufficient recaptures were comparable to estimates obtained from other methods and the relative standard errors were low (standard errors between 0.15 and 0.22). Reporting rate estimates ($0.22 - 0.26 \text{ yr}^{-1}$) were also precise (standard errors between 0.03 and 0.04) and indicated a lack of participation by the fishing industry. Estimates of natural mortality were low ($0.00 - 0.02 \text{ yr}^{-1}$) but imprecise (standard errors between 0.12 and 0.14).

3.2 Introduction

Exploitation rates of crustaceans have often been estimated by depletion methods (Delury 1947; Leslie and Davis 1939). Recently, exploitation rates in the Tasmanian rock lobster (*Jasus edwardsii*) fishery have been estimated by change-in-ratio and

index-removal methods (Frusher et al. 1997, 1998). These methods measure the relative change in abundance or composition, respectively, of the harvestable population over time. Like depletion methods, they require the population to be closed to immigration, emigration and recruitment (Pollock and Hoenig 1998; Hoenig and Pollock 1998). However, in many fisheries the population is not closed and the methods used to estimate exploitation rate must follow a specified portion of the resource over time.

Tagging studies on lobsters in northern Tasmania have shown that legal sized lobsters close to the minimum size limit molt twice a year and there is a residual amount of molting occurring throughout the year. As the population is changing throughout the year, a change in legal sized lobster abundance would relate to recruitment (molting from undersized to legal size) as well as fishing and natural mortality. Tagging studies mark and identify cohorts of animals and thus are not subject to immigration and recruitment. The effects of molting need to be eliminated so that a decline in legal sized lobster abundance relates specifically to those lobsters that were present at the start of the fishery.

Tagging studies undertaken from 1992 to 1995 had the objective of obtaining growth and movement data. Despite this, we found that the data could be used to develop and evaluate tagging models for estimating mortality rates. This paper evaluates these models.

Brownie et al. (1985) presented a number of models for estimating year specific survival and tag recovery rates from multiyear tagging studies. Klieber et al. (1987) expressed tag return data in terms of tag reporting rate and fishing and natural mortality. Hoenig et al. (1998) re-parameterized the multiyear tagging models of Brownie et al. (1985) in a very general formulation that expresses survival in terms of instantaneous rates of fishing and natural mortality. The recovery rate parameter was expressed as a product of three factors: the exploitation rate, tag reporting rate, and a parameter that includes tag loss and tag induced mortality. The exploitation rate can be expressed as a function of the fishing and natural mortality rates. Tag loss and tag induced mortality can be determined from separate studies such as aquarium and double tagging trials. Although tag reporting rate can theoretically be estimated from the models, Hoenig et al. (1998) found that tag reporting rate estimates were

unreliable unless there are many years of tagging with a wide range of fishing mortality rates.

Hearn et al. (1998) described a model that used twice a year tagging data to estimate fishing and natural mortality. They were also able to estimate tag reporting rate. Their model has a fishing period where both fishing and natural mortality affect survival and a closed period of the year where only natural mortality affects survival. Tagging is undertaken prior to and after the fishing year. We combined aspects of the models described by Hearn et al. (1998) and Hoenig et al. (1998) to best describe the dynamics of the lobster stock from the King Island area in northwest Tasmania.

3.3 Materials and Methods

3.3.1 Study design

From 1992 to 1995, lobsters were tagged during research cruises off the coast of northwestern Tasmania on the commercial fishing grounds (Figure 3.1). The same tagging sites were visited on all cruises. Lobsters were tagged immediately upon capture and released at the site of capture. Lobsters were tagged with individually numbered T-bar tags (Hallprint T-bar anchor tag; TBA1, Hallprint Pty. Ltd., 27 Jacobsen Crescent, Holden Hill, South Australia 5088, Australia). Lobsters were tagged in February, May and September from September 1992 to May 1995.

Tagging sites were situated within the main fishing grounds. Pearn (1994) found that legal sized lobsters were primarily recaptured in the location of release. For the few lobsters that did migrate, they were recovered on offshore reefs as the available habitat for *J. edwardsii* is routinely fished each season.

Notices requesting return of tags were posted to all fishers and processors known to process Tasmanian rock lobsters. To encourage participation a tag lottery was established. Each correctly completed tag return represented an entry into the lottery. Advertising of the tagging project was undertaken by regular articles in the fishing industry magazine (*Fishing Today*) and by regular oral presentations at the Tasmanian rock lobster fishermen's association meetings.

The Tasmanian rock lobster fishery opens in November for both males and females, although the exact date has varied slightly over the years from 1992 to 1995. The female and male fishing periods close at the end of April and August, respectively, of the following year. Although female lobsters are caught from May to the end of the male fishing period in August, it is mandatory that they be returned to the sea. These female lobsters are rarely checked for tags and any that were reported have not been included in the analysis. During the recapture periods, there were no reports of tagged female lobsters that died due to the fishing activity (e.g., from predation by an octopus in the trap). For application of the Hearn et al. (1998) method, September was a pre-season tagging event and May a post-season tagging event for females. Models developed in this paper utilize data on both sexes and two or three tagging events each year.

3.3.2 Development of statistical models for tagging data

To model the tag returns within a year, we need to have the fishing and natural mortality rates reflect what happens in each period between tagging events. Hoenig et al. (1998) derived formulae where fishing and natural mortality are prorated against fishing effort and the length of time since tagging, respectively. These formulae and their adaptation to our data are presented in Appendix 3.1.

3.3.3 Parameter estimability and methods for fitting models.

There is no universally applicable method for determining estimability of parameters. However, for these models the estimability can easily be demonstrated using the method of moments (Appendix 3.2).

In practice, the models are fitted by the method of maximum likelihood. The recoveries over time from each cohort of tagged animals constitute an observation from a multinomial distribution. The cohorts are considered independent. Thus the likelihood for the data is a product of the multinomials.

Models were fitted using the program SURVIV (White 1983). Because of the large number of parameters to be estimated, λ and M were held constant over all years. Program SURVIV was developed to estimate survival rates and these estimates are

constrained to be between 0 and 1. As the instantaneous mortality rates can be larger than 1, the fishing mortality parameters were transformed:

$$\hat{F} = -\ln \hat{T}$$

where \hat{F} is the fishing mortality estimate and \hat{T} is the exponential of the negative of the fishing mortality estimate. Thus, for the example in Appendix 3.1, the expected number of returns $E[R_{IBIdI}]$ is:

$$\lambda N_{IB} \exp(-0.115M)(1 - \exp(0.494 \ln T_1 - 0.271M)) \left(\frac{-0.494 \ln T_1}{-0.494 \ln T_1 + 0.271M} \right)$$

Standard errors of the fishing mortalities were obtained from the standard errors of the estimated T parameters by application of the delta method (Seber 1982). Thus,

$$s.e.(\hat{F}) = \frac{s.e.(\hat{T})}{\hat{T}}$$

3.3.4 Models tested

The available data allowed for the testing of 12 models. These models were based on two tagging strategies: tagging twice (2x, in September and May) or three times (3x, in September, February and May) per year. For each of these strategies, models were established to determine fishing mortalities for each sex separately or for sexes combined. For the latter, fishing mortality for each sex was specified as being proportional to the length of the fishing period for that sex (the length of the fishing

period for females being 60% as long as the fishing period for males.) Alternatively, fishing mortality was specified as proportional to the amount of fishing effort in the open fishing period for each sex (Table 3.1). Combining sexes is considered appropriate as both male and female lobsters are attracted to fishing traps. Fishers consider both sexes equally catchable with the exception of the period when either sex is molting. Natural mortality is also assumed to be constant over sexes.

Each of these models estimated fishing mortality annually. However, it is possible to constrain the fishing mortality to be constant over the years, and models were also evaluated with F held constant over time. Models were described using the descriptors in Table 3.2. For example, 2x_Ct_yA refers to the model based on two tagging events with fishing mortality estimated annually for sexes combined based on length of fishing period.

3.3.5 Model Selection

Initial model selection was based on parsimony. Burnham and Anderson (1998) describe parsimony as the concept that a model should be as simple as possible with respect to the model structure and number of parameters. Following the procedures of Burnham and Anderson (1998), we used Akaike's information criterion (AIC) to formally select models. This strategy compares the improvement in fit of models with increased numbers of parameters to the most parsimonious model. Only models with a substantial improvement in fit are considered more appropriate than the parsimonious model.

3.3.6 Diagnostics for lack of fit

Recently Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062) suggested looking at residuals as a diagnostic tool for assessing tagging models. We analyzed the residuals of the selected model for patterns that would indicate the problems described by Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062).

3.4 Results and Discussion

The number of legal sized lobsters tagged in the survey periods ranged from 94 to 577 lobsters; the number of recaptures from the tagged cohorts ranged from 6 to 129 (Table 3.3).

3.4.1 Model selection

Data for all models were found to be overdispersed and quasi-likelihood adjustments of the AIC values were undertaken (see Burnham and Anderson 1998) (Table 3.4)

Two groups of models used equivalent amounts of data and model selection within these groups was based on their AIC weights (Burnham and Anderson 1998) (Table 3.4). The 2x group was based on considering two tagging events per year (September and May) and the 3x group was based on three tagging events per year (September, February and May). In the 3x group models 3x_Ce_yA, 3x_S_yC and 3x_Ce_yC had support values less than 2 as had models 2x_Ct_yA and 2x_Ce_yA in the 2x group (Table 3.4). These models should be considered as plausible models for the northwest Tasmanian tag recapture data.

Common to both groups was the Ce_yA model which estimates annual fishing mortalities with sexes combined based on effort. The other model supported in the 2x group was the model estimating annual fishing mortalities with sexes combined based on the duration of the fishing years. This is not surprising because when the February tagging event is eliminated the fishing period for males is reduced from 3 (Nov-Feb, Feb-May, May-Aug) to 2 (Nov-May, May-Aug) periods and from 2 (Nov-Feb, Feb-Apr) to 1 (Nov-Apr) period for females (Table 3.1). Although a model based on separate sexes was given support in the 3x group, the addition of the extra parameter (sex effect) did not have a substantial improvement over the most parsimonious model and this model is not considered further.

In the 3x model group, fishing mortality estimates with sexes combined based on length of the fishing periods were poorly supported indicating that the extra tagging period (February) split a period when fishing effort was disproportional to time. There are differences in effort between tagging periods with the period between the second and third tagging surveys (February to May) having the greatest effort expended per

unit of time (Figure 3.2). In contrast, the period between the third and first tagging surveys of the following year (May to September) had less effort per unit of time. Over the survey periods, the May to September period has shown a steady increase in the amount of effort expended per unit of time which is reflecting the increase in price paid over time for lobster during this period (Frusher 1997). Effort is considered the most appropriate means of partitioning fishing mortalities within a fishing year.

The models producing annual estimates of fishing mortalities with fishing mortality within a year based on fishing effort were considered the most appropriate models for describing the dynamics of the lobster stocks in northwestern Tasmania. As the 2x_Ce_yA and 3x_Ce_yA models used different amounts of data, model selection was based on the relative standard errors of the parameters being estimated. In all cases the model based on three tagging events per year had lower relative standard errors than the model based on two tagging events per year.

3.4.2 Model fit

The residuals from the fitted models (e.g., Table 3.5) display none of the symptoms of problems described by Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062) (i.e. row effects and diagonal effects). There is a preponderance of negative residuals, however, and some columns contain residuals predominantly of the same sign. Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062) were unable to generate this kind of column pattern in their simulations though they specifically tried to produce this effect (R. Latour, VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062). The model considered here is more complex than the model of Hoenig et al. (1998) which was investigated by Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062). Thus there are more ways that problems could occur. For example, if the residuals for females were of the opposite sign as those of males in a particular recovery year that might suggest that the molting season for females was unusually early or late relative to molting of males in that year. This does not appear to be the case here. Like Latour (VIMS, PO Box 1346 Gloucester Point, Virginia, USA 23062), we are unable to suggest a mechanism that would generate a column effect. We suggest that this may be an important area for research.

A reviewer of the manuscript suggested that certain cells of the recovery matrix with large residuals may be exerting an undue influence on the estimates. We examined the table of residuals (Table 3.5) for evidence of a problem and tried fitting the model after deleting cells for which the residuals were large (> 5). After deleting these cells, there were only minor changes in the parameter estimates. Exploitation rates for the 1991/1992 to 1993/1994 fishing years increased from 64% to 67% and declined for the 1994/1995 (71% to 65%), 1995/1996 (63% to 52%) and 1996/1997 (37% to 18%) fishing years. The larger differences for the last two fishing years are considered to be due to the low number of tags returned after tagging ceased in May 1995 as previously described. Tag reporting rate increased from 22% to 24% and natural mortality remained at zero. The precision of all estimates decreased after the cells with large residuals were removed and there was no change in the column effects of the residuals.

We conclude that although there are some patterns in the residuals, it is unclear what the cause may be and it is also unclear how the unidentified problem might bias the results. However, as discussed below, the overall results are consistent with those results obtained from the assessment model of Punt and Kennedy (1998) so there is reason to believe that the model has provided reasonable results.

3.4.3 Fishing mortality

Tagging during this study commenced part way through the 1991/1992 fishing year in May 1992. The female component of this fishing year had already closed and thus fishing mortality estimates are based on males for 1991/1992. Only 17 percent of the fishing effort for the 1991/1992 fishing year was expended between May and September 1992 and thus male fishing mortality for this period is based on recaptures from limited effort.

Comparison of the relative standard error (Rse – standard error divided by estimate) of the supported models, which estimated annual fishing mortalities, showed that the 3x_Ce model provided the most precise estimates for all fishing years (Figure 3.3 a).

The fishing mortality estimates for the 1991/1992 to 1993/1994 fishing years were similar. Fishing mortality estimates for 1994/1995 to 1996/1997 were greater for the

3x than 2x models. The 2x model showed a decline in fishing mortality estimates from 1993/1994 whereas this was only apparent in the 1996/1997 fishing year for the 3x model.

Fishing effort is often used as a measure of relative fishing mortality. Fishing effort for the region where tagging was undertaken is available from 1992 onwards. Trends in fishing effort are comparable to the fishing mortality estimates from both models although the 3x model appears a better relative comparison (Figure 3.4). Only in the 1996/1997 fishing year was the trend in effort substantially different to the trend in annual fishing mortality estimates whereas the last two fishing years diverge from the effort trend in the 2x model.

The precision (equivalent to Rse) of the estimates for the most recent years decline with the 1996/1997 estimates being approximately half as precise as the earlier fishing year estimates (with the exception of the 1991/1992 fishing year when the estimate was based on 17% of the fishing effort).

The declining precision for the last fishing years is likely to be due to the low number of tags returned during this period as tagging ceased in May 1995 and exploitation rates are high. In the fishing year (1995/1996) immediately after tagging ceased, 60% of the tagging event-recapture period strata had 2 or less returns. In the 1996/1997 fishing year this increased to 98% with 66% having zero returns (Table 3.3).

The most precise estimates were obtained from the 3x_Ce_yA model. These estimates are high with annual harvesting of between 60 and 70 percent of the legal sized biomass for all fishing years except the last (Table 3.6). Although high, these estimates are similar to those found by Frusher et al. (1998) for other regions of Tasmania and have similar trends to biomass estimates obtained from a rock lobster assessment model (Punt and Kennedy 1998) with the exception of the final year. The low fishing mortality estimate for the final year is considered to be due to the low expected rate of tag returns as discussed earlier.

High exploitation rates in the Tasmanian rock lobster fishery are not unexpected as Frusher (1997) reports legal sized biomass to be around 6% of the virgin legal sized biomass. Additionally, the majority of the legal sized biomass is comprised of recent recruits (Frusher 1997).

3.4.4 Tag reporting rate

In the models tested, tag reporting rate was held constant over all years. Although fishers' behavior with respect to reporting tags can vary between years, there were no major fishery policy changes during the period of surveys to have altered fishers' behavior.

Reporting rate and its relative precision showed little variation between models and was estimated reasonably precisely (Figure 3.3 b). Frusher (pers. observ.) found no tag loss or tag induced mortality from over 100 lobsters held in aquaria. Treble (1995) found tag loss to be approximately 3% annually for T-bar tags used in the same species. Although these components of the reporting rate parameter are considered to be low, reporting rate estimates were low (0.20 to 0.23) indicating that fisher cooperation in looking for tags and subsequently reporting tags was poor.

3.4.5 Natural mortality

Natural mortality was also held constant over all years. Natural mortality estimates were very close to zero or very low, ranging from 0 - 0.018 yr⁻¹ (Figure 3.3 c). However, the standard errors were relatively large. This indicates that the models tested had trouble estimating this parameter. This may be due to the relatively short period of time that natural mortality was responsible for total mortality (i.e. when the fishery was closed). In the Tasmanian fishery, the female fishing period is closed for approximately half the year. Hearn et al. (1998) expected that their method would work well in a pulse fishery (i.e., when the fishery is open for such a short time that natural mortality can be considered to be negligible during the fishing period) or when applied to a fishery with a short fishing period (say, less than four months).

A concern with the very low estimates of M is that they could be highly correlated with tag reporting rate estimates. SURVIV (White 1983) generates a covariance matrix, which found estimates of M not to be highly correlated with estimates of reporting rate. We confirmed this by re-running the model with M fixed at 0.1, 0.15 and 0.2. These resulted in an increase in tag reporting rate of 11, 18 and 25% respectively and a decrease in F 's by 10, 15 and 20% respectively for the first three fishing seasons. These variations were smaller for the later years of the study.

Natural mortality has proved very difficult to estimate in lobster stocks. Kennedy (1992) tried to estimate natural mortality of female lobsters from long term (> 15 years) recaptures. Although no definitive estimate could be made from the data, he considered that natural mortality for the southern Tasmanian females to be 0.1 or less. Hearn et al. (1998) found that their method produced accurate estimates of M when M was high, although large numbers of animals were required to be tagged (> 1000) when fishing mortality was low and reporting rate was 50% or less.

Low tag reporting rates and high fishing mortality rates would make a low natural mortality estimate difficult from tag-recapture data. The high fishing mortality will result in few tags being available after one period of fishing and, combined with low tag reporting rate, few tags will be available to estimate M . Frusher and Hoenig (unpublished data) found that improved precision in the estimate of M was best achieved by either improving the tag reporting rate or increasing the duration of the tagging events from three to six fishing seasons.

The large standard errors of the natural mortality estimate also indicate that natural mortality is difficult to determine. However, the natural mortality estimates from all models tested were low which supports the inferences made by Kennedy (1992).

This is the first time these models have been applied to a real situation and the results are encouraging. Fishing mortality estimates had reasonable estimated precision, and the trends matched results from other methods. Although it is possible to estimate fishing mortality in years after the cessation of tagging, this does not appear to work well when fishing mortality is high due to the tags being “fished out”. It is well known that most methods that produce estimates of fishing and natural mortality tend to produce highly correlated estimates. But, the estimated correlations in this study were modest.

This analysis is also the first time that a detailed model with multiple tagging events within a fishing year has been tested. The 3x model with fishing mortality apportioned by fishing effort and with sexes combined appears appropriate for estimating fishing mortality in the northern Tasmanian rock lobster fishery.

While the precision of natural mortality estimates was low, all models indicated that natural mortality was low.

Despite the poor reporting rate, relatively precise estimates of fishing mortality could still be achieved. This is important for the northern Tasmanian fishery where attempts at estimating fishing mortality rate using other methods have been compromised by the frequent molts and extended molting period of the lobsters.

Table 3.1. Annual proportion of time based on fishing effort (trap lifts) and time (days) used to calculate fishing and natural mortality respectively.

Fishing year	Units	Sex	Periods used in analysis (see text for details)					
1991/ 1992	Dates						11 May – 31 Aug.	1 Sept.-19 Sept.
	Trap lifts	M					0.17	0
		F					0	0
	Day						0.31	0.052
1992/ 1993	Dates		20 Sept. – 31 Oct.	1 Nov. – 6 Feb.	7 Feb. – 31 Apr.	1 May – 10 May	11 May – 31 Aug.	1 Sept.-19 Sept.
	Trap lifts	M	0	0.392	0.402	0.029	0.177	0
		F	0	0.494	0.506	0	0	0
	Day		0.115	0.271	0.225	0.027	0.31	0.052
1993/ 1994	Dates		20 Sept. – 17 Nov.	18 Nov. – 13 Feb.	14 Feb. – 31 Apr.	1 May – 10 May	11 May – 31 Aug.	1 Sept.-19 Sept.
	Trap lifts	M	0	0.379	0.366	0.026	0.228	0
		F	0	0.509	0.491	0	0	0
	Day		0.164	0.238	0.208	0.027	0.31	0.052
1994/ 1995	Dates		20 Sept. – 17 Nov.	18 Nov. – 29 Jan.	30 Jan. – 31 Apr.	1 May – 22 May	23 May – 31 Aug.	1 Sept.-19 Sept.
	Trap lifts	M	0	0.235	0.422	0.082	0.261	0
		F	0	0.358	0.642	0	0	0
	Day		0.164	0.197	0.249	0.06	0.277	0.052
1995/ 1996	Dates		20 Sept. – 30 Nov.	1 Dec. – 6 Feb.	7 Feb. – 31 Apr.	1 May – 10 May	11 May – 31 Aug	1 Sept.-19 Sept.
	Trap lifts t	M	0	0.268	0.415	0.041	0.275	0
		F	0	0.392	0.608	0	0	0
	Day		0.197	0.189	0.227	0.027	0.309	0.052
1996/ 1997	Dates		20 Sept. – 17 Nov.	18 Nov. – 6 Feb.	7 Feb. – 31 Apr.	1 May – 10 May	11 May – 31 Aug.	1 Sept.-19 Sept.
	Trap lifts	M	0	0.308	0.318	0.023	0.351	0
		F	0	0.492	0.508	0	0	0
	Day		0.164	0.222	0.225	0.027	0.310	0.052

Table 3.2. Terminology used to describe models.

Model code	Description
2x	Two tagging events per year (September and May)
3x	Three tagging events per year (September, February and May)
S	F 's estimated for separate sexes
Ct	F 's estimated for sexes combined based on lengths of time of fishing periods
Ce	F 's estimated for sexes combined based on amount of fishing effort in each sex's fishing period
yA	F 's estimated for each year of recaptures
yC	F 's estimated for all years combined (constant F)

Table 3.3. Tag and recapture details of male and female legal sized lobsters from May 1992 to September 1997.

Year/ timing	Tag date	Sex	No. Tag	Recapture period															
				1992 Sept.	Feb.	1993 May	Sept.	Feb.	1994 May	Sept.	Feb.	1995 May	Sept.	Feb.	1996 May	Sept.	Feb.	1997 May	Sept.
1a	May/ 1992	M	333	13	26	23	5	4	6	1	0	1	2	0	0	0	0	0	0
		F	220		20	12		8	2		1	1		1	0		1	0	
2b	Sept. /1992	M	486		53	27	5	15	13	6	4	2	1	0	1	2	0	0	0
		F	577		36	22		19	14		12	3		1	2		2	0	
2d	Feb. /1993	M	280			15	6	6	10	1	9	2	1	1	0	2	0	0	0
		F	240			10		16	7		11	7		0	0		1	1	
2a	May /1993	M	181				3	21	9	0	4	3	1	3	0	0	0	0	1
		F	105					17	8		5	1		4	1		0	2	
3b	Sept. /1993	M	226					15	6	1	6	3	9	1	1	0	1	0	0
		F	228					21	10		4	5		4	2		0	0	
3d	Feb. /1994	M	357						22	0	17	6	6	2	2	3	0	0	0
		F	323						14		21	6		7	4		1	2	
3a	May /1994	M	229							5	16	13	8	2	5	3	0	1	0
		F	154								11	5		2	2		1	0	
4b	Sept. /1994	M	168								12	16	5	2	3	2	0	0	1
		F	224								8	11		7	2		1	0	
4d	Jan. /1995	M	267									17	6	8	3	10	1	0	0
		F	230									19		7	5		12	1	
4a	May /1995	M	159										8	11	7	11	0	0	2
		F	94											3	3		0	0	

Where a = after female fishing year, b = before start of male and female fishing year and d = during female and male fishing year.
Blank spaces in the recapture part of the table indicate that no fishing occurred for that sex for that period in time.

Table 3.4. Dispersion coefficients (\hat{c}), $\Delta QAIC$ values, AIC weights ($wQAIC$), and level of support for two groups of models based on two (2x) and three (3x) tagging events per year. A $\Delta QAIC$ value is the difference between the QAIC value for a model and the smallest QAIC value for any model in the group (see Burnham and Anderson 1998 for further details). See Table 3.2 for model descriptors. Models with support values less than 2 are considered plausible models.

Group	Model	\hat{c}	$\Delta QAIC$	$wQAIC$	Support
3x	S_yA	2.05	3.02	0.07	4.53
	S_yC	2.15	0.10	0.29	1.05
	Ct_yA	2.14	4.35	0.03	8.81
	Ct_yC	2.18	4.32	0.03	8.69
	Ce_yA	2.11	0.00	0.30	1.00
	Ce_yC	2.15	0.19	0.27	1.10
2x	S_yA	1.92	4.12	0.06	7.84
	S_yC	2.41	5.57	0.03	16.23
	Ct_yA	1.95	0.00	0.43	1.00
	Ct_yC	2.20	3.08	0.09	4.67
	Ce_yA	1.96	0.52	0.33	1.30
	Ce_yC	2.20	3.90	0.06	7.06

Table 3.5. Recapture residuals (observed – expected recaptures) of male and female legal sized lobsters from May 1992 to September 1997 for the 3x_Ce_yA model. Negative residuals are in bold.

Year/ timing	Tag date	Sex	No. Tagged	Recapture period															
				1992 Sept. ¹	Feb. ²	1993 May ³	Sept.	Feb.	1994 May	Sept.	Feb.	1995 May	Sept.	Feb.	1996 May	Sept.	Feb.	1997 May	Sept.
1a	May	M	333	1.3	5.5	8.3	0.6	-3.0	1.1	-1.1	-2.0	-1.7	1.1	-0.5	-0.6	-0.3	-0.1	-0.1	-0.1
	/1992	F	220		3.9	1.1		1.2	-2.5		-1.5	-2.0		-0.0	-1.1		0.7	-0.3	
2b	Sept.	M	486		17.5	1.4	-2.6	2.8	4.5	2.4	0.6	-2.7	-0.5	-0.9	-0.1	1.5	-0.2	-0.2	-0.2
	/1992	F	577		-6.2	-6.7		1.1	2.2		5.4	-4.9		-1.7	-1.0		1.2	-0.7	
2d	Feb.	M	280			-7.1	-0.6	-4.6	2.6	-2.1	6.0	-2.1	-0.3	0.2	-1.0	1.6	-0.2	-0.2	-0.1
	/1993	F	240			-7.9		4.8	-0.4		6.9	2.1		-1.7	-1.9		0.5	0.6	
2a	May	M	181				-3.6	10.4	1.6	-3.1	1.0	-1.1	-0.3	2.2	-1.0	-0.4	-0.2	-0.2	0.9
	/1993	F	105					9.6	3.1		2.3	-2.3		2.9	-0.2		-0.3	1.7	
3b	Sept.	M	226					-1.0	-5.1	-3.7	1.5	-3.2	7.0	-0.2	-0.4	-0.6	0.7	-0.2	-0.2
	/1993	F	228					4.9	-0.6		-1.9	-2.1		1.6	-0.7		-0.7	-0.6	
3d	Feb.	M	357						-3.92	-10.9	6.5	-8.4	1.4	-0.8	-1.4	1.6	-0.6	-0.6	-0.5
	/1994	F	323						-8.2		8.7	-8.8		1.9	-1.6		-0.5	0.7	
3a	May	M	229							-5.5	5.9	-0.8	3.6	-0.7	1.8	1.7	-0.6	0.5	-0.5
	/1994	F	154								2.4	-5.3		-1.5	-1.9		-0.0	-0.9	
4b	Sept.	M	168								2.7	3.2	0.9	-0.5	0.0	0.8	-0.5	-0.5	0.6
	/1994	F	224								-4.4	-4.0		1.9	-3.6		-0.5	-1.3	
4d	Jan.	M	267									-10.3	-2.7	2.7	-3.3	7.4	-0.1	-1.1	-0.9
	/1995	F	230									-1.6		-0.0	-2.8		10.0	-0.8	
4a	May	M	159										-1.7	5.1	-0.1	8.1	-1.3	-1.2	1.0
	/1995	F	94											-1.8	-2.4		-1.4	-1.3	

Where a = after female fishing year, b = before start of male and female fishing year and d = during female and male fishing year.

Blank spaces in the recapture part of the table indicate that no fishing occurred for that sex for that period in time.

¹Refers to the period from mid May to mid Sept. Actual dates are given in Table 3.1.

²Refers to the period from mid-Sept. to mid-Feb.. Actual dates are given in Table 3.1.

³Refers to the period from mid-Feb. to mid-May. Actual dates are given in Table 3.1.

Table 3.6. Instantaneous fishing mortality rates (F , yr^{-1}), exploitation rates ($u = 1 - e^{-F}$), annual catches (C) and estimated biomass ($B = C/u$) from the 3x_Ce_yA model.

Fishing year (Nov. to Aug.)	F	u	C (tonnes)	B (tonnes)
1991/1992	1.02	0.64	431.7	674.5
1992/1993	1.03	0.64	364.8	566.5
1993/1994	1.03	0.64	392.2	611.9
1994/1995	1.24	0.71	362.1	509.2
1995/1996	1.00	0.63	360.4	571.1
1996/1997	0.47	0.38	325.8	866.6

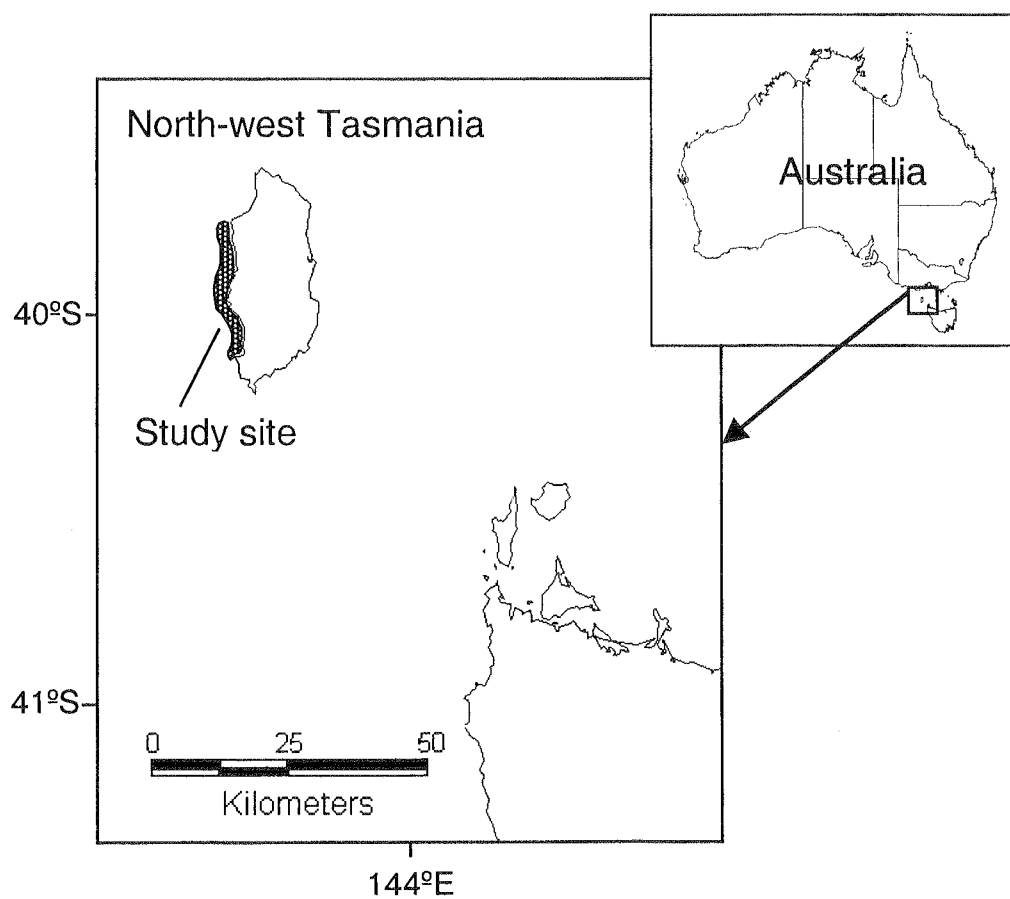


Figure 3.1. Map of Australia showing north west Tasmania and study site.

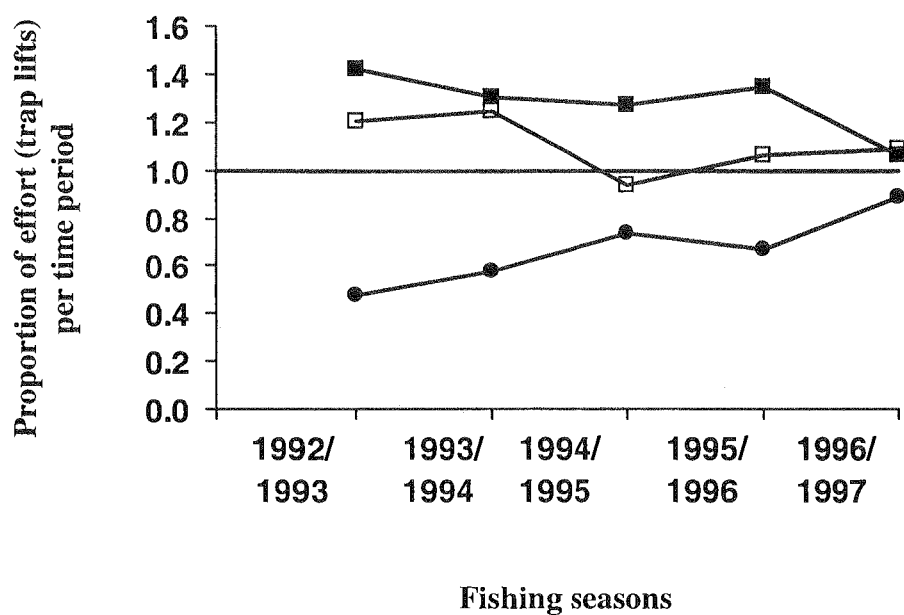


Figure 3.2. Comparison of fishing effort expended between September and February (open squares), February to May (closed squares) and May to September (solid circles) for the 1992/1993 to 1996/1997 fishing years. The horizontal line shows where the proportion of effort is equivalent to the proportion of time.

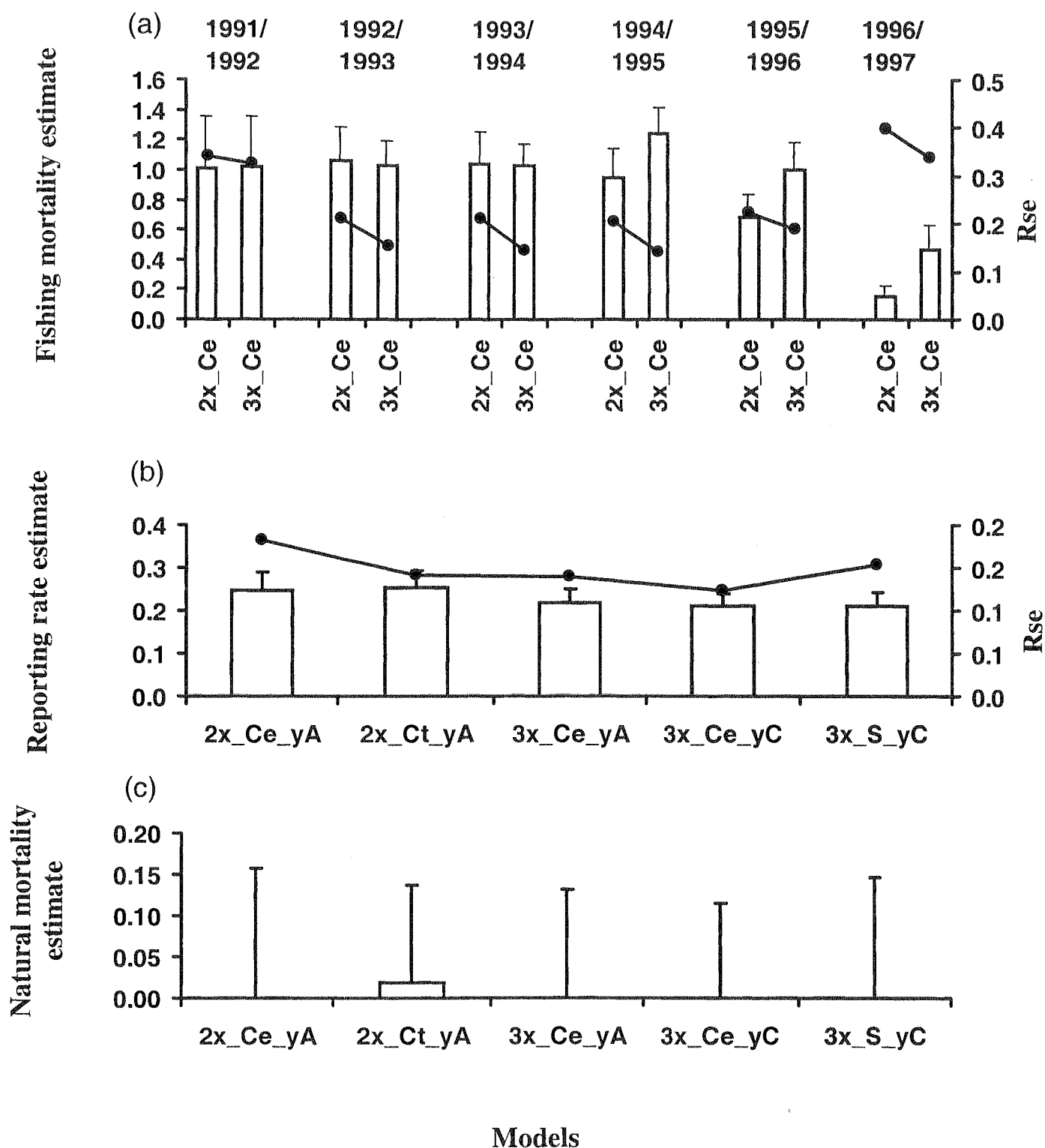


Figure 3.3. Comparison of instantaneous fishing mortality rate (a), tag reporting rate (b) and instantaneous natural mortality rate (c) estimates from 2x and 3x models. Bars give estimated parameters with one standard error. Connected dots show relative standard error (Rse). Meaningful relative standard error estimates can not be obtained for natural mortality estimates as they approached zero.

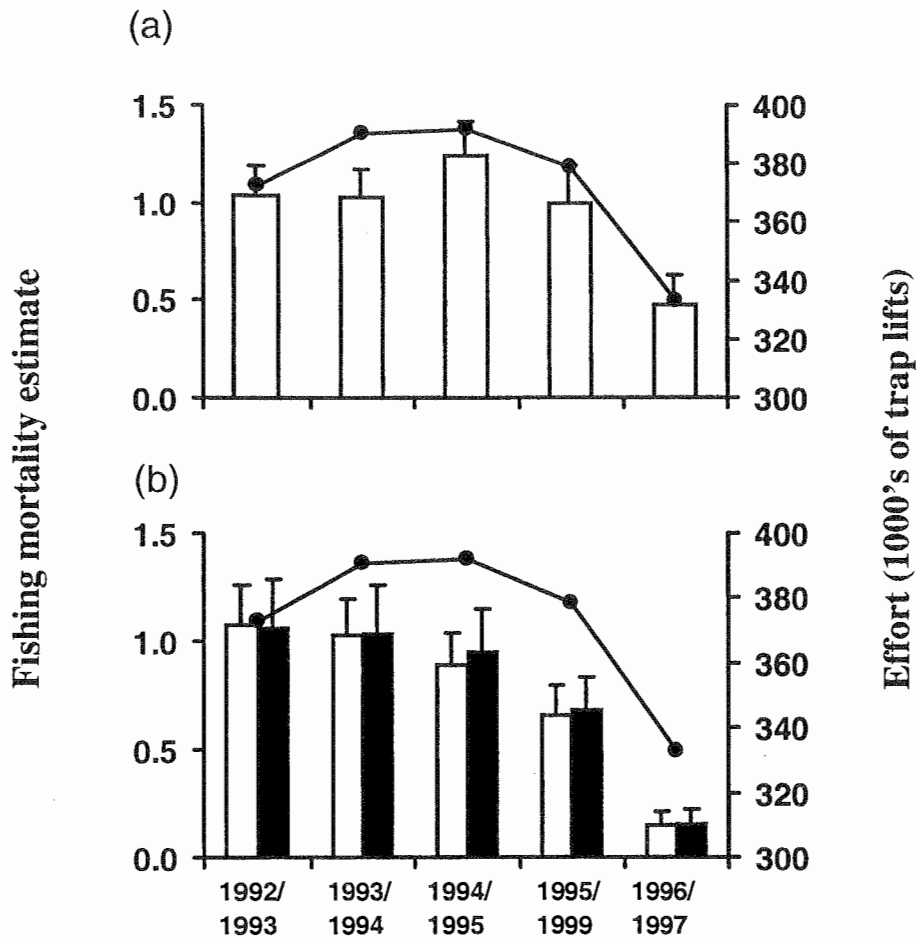


Figure 3.4. Comparison of annual fishing effort and fishing mortality estimates from the 3x model (a) and the 2x model (b) based on time (light) and effort (dark). Bars give estimated fishing mortality with one standard error. Connected dots show effort.

Appendix 3.1: Derivation of exploitation rate in terms of natural and fishing mortality components and examples of application to a model using three tagging periods.

The notation used is as follows:

λ = a composite parameter that represents the joint probability of three events: the probability that a tag will be found and reported to the fisheries biologist, given that the tagged lobster has been harvested; the probability that a tagged lobster does not die from the tagging process; and the probability that a tag is not shed immediately. λ is assumed constant over time.

F_i = instantaneous fishing mortality rate in year i .

F_{mki} = instantaneous male fishing mortality in period k of year i .

F_{fki} = instantaneous female fishing mortality in period k of year i .

M = instantaneous natural mortality rate. M is assumed constant over time and equal for both sexes.

N_{mhi} = number of male lobsters tagged and released at time h of year i .

N_{fhi} = number of female lobsters tagged and released at time h of year i .

ε_{ih} = fraction of the annual fishing effort expended in year i during period h .

Δt_h = length of the h th period of the year.

$u_i(F_i, M)$ = exploitation rate in year i . For convenience, this will be abbreviated to u_i when clarity is not affected.

R_{ihjk} = number of recaptures of animals tagged in year i at time h and recaptured in year j during period k .

$E[R_{ihjk}]$ = expected value of R_{ihjk} .

Let the year be divided into $K = 3$ periods. The exploitation rate in year i can be expressed as:

$$u_i(F_i, M) = \sum_{k=1}^{K=3} a_{ik} b_{ik} c_{ik}$$

Here, a_{ik} is the fraction of the population surviving to the beginning of period k , with $a_{i1} = 1$ and

$$a_{ik} = \exp\left(-M \sum_{h=1}^{k-1} \Delta t_h - F_i \sum_{h=1}^{k-1} \varepsilon_{ih}\right), k > 1$$

The a_{ik} term prorates the natural mortality according to how much of the year has elapsed since tagging, and prorates the annual fishing mortality according to how much of the fishing effort has occurred up to the start of the k^{th} period. The b_{ik} term is the fraction dying in period k

$$b_{ik} = 1 - \exp(-M\Delta t_k - F_i \varepsilon_{ik}),$$

and c_{ik} is the fraction of the deaths in period k due to fishing

$$c_{ik} = \frac{F_i \varepsilon_{ik}}{F_i \varepsilon_{ik} + M\Delta t_k}$$

The following example is based on female lobsters in northwest Tasmania and partitions a year based on fishing periods and tagging events (Table 3A.1). The tagging events are before (B), during (D) and after (A) the fishing year. Fishing periods divide the year into before (b), during (d1 and d2) and after (a1 and a2) the fishing year. The portions of effort and time are obtained from Table 3.1. Because recaptures are dependent on the fishery, recaptures are only obtained in period's d1 and d2.

The expected recoveries in the first portion of the fishing year (d1) in year 1 from tagging before the fishing year (B) in year 1 is:

$$E[R_{1B|d1}] = \lambda N_{1B} \exp(-0.115M)(1 - \exp(-0.494F_1 - 0.271M)) \left(\frac{0.494F_1}{0.494F_1 + 0.271M} \right)$$

where N_{1B} is the number of lobsters tagged at time B, $\exp(-0.115M)$ is the number of lobsters which have survived period b (of length 0.115 yr),

$(1 - \exp(-0.494F_1 - 0.271M))$ is the number of lobsters dying in period d1, and

$\left(\frac{0.494F_1}{0.494F_1 + 0.271M} \right)$ is the proportion of lobsters dying in period d1 from fishing.

The expected recoveries in the second portion of the fishing year (d2) in year 1 from tagging before the fishing year (B) in year 1 is:

$$E[R_{1B|d2}] =$$

$$\lambda N_{1B} \exp(-0.386M - 0.496F_1)(1 - \exp(-0.506F_1 - 0.225M)) \left(\frac{0.506F_1}{0.506F_1 + 0.225M} \right)$$

where $\exp(-0.386M - 0.496F_1)$ is the number of lobsters which have survived prior to the start of period d2 (i.e. survived period b and d1), $(1 - \exp(-0.506F_1 - 0.225M))$ is

the number of lobsters dying in period d2, and $\left(\frac{0.506F_1}{0.506F_1 + 0.225M} \right)$ is the proportion of lobsters dying in period d2 from fishing.

From the second tagging event (D) in year 1 there would only be recaptures in year 1 from period d2. The expected recovery is

$$E[R_{1D1d2}] = \lambda N_{1D} (1 - \exp(-0.506F_1 - 0.225M)) \left(\frac{0.506F_1}{0.506F_1 + 0.225M} \right)$$

Note that there is no term for lobsters surviving prior to exploitation of the fishery as tagging commenced within the fishing year.

There are no recoveries in year 1 from tagging after the fishing year (A).

In year 2 there would be expected recoveries from each of the tagging events in year 1 and, depending on the time of tagging, from some tagging events in year 2.

The expected recoveries in period d1 of year 2 from tagging events B, D and A of year 1 and tagging event B of year 2 are $E[R_{1B2d1}]$, $E[R_{1D2d1}]$, $E[R_{1A2d1}]$ and $E[R_{2B2d1}]$, respectively, and are as follows:

$$E[R_{1B2d1}] =$$

$$\lambda N_{1B} \exp(-1.164M - F_1) (1 - \exp(-0.509F_2 - 0.238M)) \left(\frac{0.509F_2}{0.509F_2 + 0.238M} \right)$$

$$E[R_{1D2d1}] =$$

$$\lambda N_{1D} \exp(-0.778M - 0.506F_1) (1 - \exp(-0.509F_2 - 0.238M)) \left(\frac{0.509F_2}{0.509F_2 + 0.238M} \right)$$

$$E[R_{1A2d1}] = \lambda N_{1A} \exp(-0.526M) (1 - \exp(-0.509F_2 - 0.238M)) \left(\frac{0.509F_2}{0.509F_2 + 0.238M} \right)$$

$$E[R_{2B2d1}] = \lambda N_{2B} \exp(-0.164M) (1 - \exp(-0.509F_2 - 0.238M)) \left(\frac{0.509F_2}{0.509F_2 + 0.238M} \right)$$

Table 3A.1. Terminology of fishing and tagging periods used in the text and example of apportionment of effort and time for two fishing years for female lobsters. The two years correspond to the 1992/1993 and 1993/1994 fishing years of Table 3.1.

Year	Apporti- onment	Tag (Sept.)	Fishing closed (Sept.-Nov.)	Fishing open (Nov.-Feb.)	Tag (Feb.)	Fishing open (Feb.-Apr.)	Fishing closed (May)	Tag (May)	Fishing closed (May-Sept.)
period		B	b	d1	D	d2	a1	A	a2
1	effort		0	0.494		0.506	0		0
	time		0.115	0.271		0.225	0.027		0.362
2	effort		0	0.509		0.491	0		0
	time		0.164	0.238		0.208	0.027		0.362

Appendix 3.2: Parameter estimability.

In the example in Appendix 3.1, an estimate of M can be obtained by dividing

$E[R_{1A2d1}]$ by $E[R_{2B2d1}]$ which yields : $\frac{N_{1A} \exp(-0.526M)}{N_{2B} \exp(-0.164M)}$ and can be equated with

the observed ratio of recaptures R_{1A2d1}/R_{2B2d1} . This is easily solved to obtain an estimate of M . Similarly, dividing $E[R_{1B2d1}]$ by $E[R_{1A2d1}]$ or $E[R_{2B2d1}]$ yields :

$$\frac{N_{1B} \exp(-F_1 - 1.164M)}{N_{1A} \exp(-0.526M)} = R_{1B2d1}/R_{1A2d1} \text{ or } \frac{N_{1B} \exp(-F_1 - 1.164M)}{N_{2B} \exp(-0.164M)} = R_{1B2d1}/R_{2B2d1} ,$$

either of which is easily solved for F_1 once M is known. Once M and F_1 are known, λ can be obtained from $E[R_{1B1d1}] = R_{1B1d1}$. Similar logic can be used to establish the estimability of all of the parameters in the models considered here.

4. Strategies for improving the precision of fishing and natural mortality estimates from multiyear tagging models: a case study

(This chapter has been accepted for publication in the Journal of Marine and Freshwater Research)

4.1 Abstract.

Fisheries scientists and managers want precise estimates of fishing and natural mortality for assessments and management decisions. Because tagging can be an expensive option, maximizing potential benefits requires careful consideration of experimental design. We evaluated four options for improving the precision of estimates by conducting Monte Carlo simulations of a fishery patterned after that for the rock lobsters in north-western Tasmania, Australia. These options were increasing the number of animals tagged each tagging event, increasing the frequency of tagging events each year, improving tag reporting rate and increasing the duration of the study. The latter option considered only a design based on twice a year tagging for 3 years followed by once a year tagging for three years to minimise costs. Improving tag-reporting rate and increasing the duration of the tagging study provided greater improvements in precision than increasing either the number of lobsters tagged per tagging event or the number of tagging events each year. A design based on twice-a-year tagging for three years to determine tag-reporting rate followed by an additional two or more years of once-a-year tagging provided the most precise estimates of natural mortality and was cost effective relative to the other options. However, tag-reporting rate must be constant over the period of the study. If constancy cannot be assumed, then years of multiple tagging events may have to be interspersed among the years with single tagging events. Further improvements in mortality estimates from this multiyear tagging model could be achieved with improved tag-reporting rate.

4.2 Introduction

Managers and scientists want the most precise estimates possible of fishing mortality rates for their assessments, but financial and logistical constraints limit what can be achieved. Precise estimates permit greater resolution of changes in annual fishing mortalities so that fishing activity can be managed rationally. As fishing and natural mortality rates are an important component of many fishery assessments, the precision of these estimates strongly affects the precision of assessment outcomes.

Frusher and Hoenig (in press) obtained estimates of reporting rate and fishing and natural mortality rates from multiyear tagging models for rock lobsters in north-west Tasmania. Improving the precision of these estimates usually comes at a cost, which could involve either tagging more lobsters or improving publicity campaigns to increase tag-reporting rates. Xiao (1996) highlighted the need to evaluate the experimental design to ensure that sufficient animals are tagged to produce meaningful results. Conversely, tagging too many animals can lead to precision beyond what is required by management and thus be wasteful of both labour and funds. Changes in the study design and the rewards can also improve precision at no, or very little, additional cost (Nichols *et al.* 1991; Pollock *et al.* in press).

The present paper evaluates four scenarios that could be used to improve the relative standard errors (*Rse*'s) of fishing and natural mortality estimates obtained from multiyear tagging studies of rock lobsters.

4.3 Methods

The models used in this paper are derived from the multiyear tagging models of Brownie *et al.* (1978, 1985) as extended by Hoenig *et al.* (1998) and Hearn *et al.* (1998). The expected recoveries from a Brownie model are presented in Table 4.1. The survival-rate parameters (S_i) can be expressed in terms of instantaneous fishing and natural mortality rates as follows:

$$(1) \quad S_i = \exp((-F_i - M_i)\Delta t_i)$$

where F_i = instantaneous fishing mortality rate (year^{-1}) in time period i and M_i = instantaneous natural mortality rate (year^{-1}) in time period i . In these simulations M is assumed constant over time and equal for the two sexes. Consequently the subscript is suppressed. Δt_i = length of the i th time period (year).

The tag-recovery rates (f_i) are also functions of the F 's and M , but the functional form depends on the relative timing of the fishing and natural mortalities. If both occur with constant intensity over the course of a period (Type 2 fishery of Ricker 1975), then the tag-recovery rate can be expressed as

$$(2) \quad f_i = \lambda_i (1 - \exp((-F_i - M)\Delta t_i)) \left(\frac{F_i}{F_i + M} \right)$$

where λ_i is a composite parameter that represents the joint probability of three events: a tagged lobster does not die from the tagging process; a tag is not shed immediately; and a tag will be found and reported to the fisheries biologist, given that the tagged lobster has been harvested. In these simulations λ is assumed constant over time.

For example, the expected recoveries in period 2 from tagging at event 1 is $N_1 S_1 f_2$ (Table 4.1). Substituting equations 1 and 2 for S_1 and f_2 , respectively, we can write the expected recoveries in terms of F_1 , F_2 , and M

$$(3) \quad N_1 S_1 f_2 = N_1 \exp((-F_1 - M)\Delta t_1) \lambda (1 - \exp((-F_2 - M)\Delta t_2)) \left(\frac{F_2}{F_2 + M} \right)$$

Frusher and Hoenig (in press) analysed data from a tagging study in which lobsters were tagged three times per year, in September (before the beginning of the fishing season), February (mid-season), and May (end of female season). The male season continues past the end of the female season until August. Rather than estimating separate fishing mortality rates for each sex, they used the following approach.

- (1) A single fishing mortality rate for males was estimated per year and was apportioned to the period of the year according to the relative amount of fishing effort occurring in the periods.
- (2) Because the fishing season for females was contained within the season for males, the fishing mortality for females in a year was assigned to be a fraction of the male mortality; the fraction was equal to the amount of fishing effort in the female season divided by the total effort in the year.
- (3) Natural mortality, M , was assumed equal for the sexes and held constant over years. It was apportioned to periods of the year according to the lengths of the periods.

A more detailed description of the tagging models can be found in Frusher and Hoenig (in press). Because conducting three tagging events per year may be logistically or otherwise impossible, Frusher and Hoenig also considered what results would be achieved if only two tagging events per year occurred, in September and May. Two or more tagging events per year are required to estimate the reporting rate and so allow the separation of the total mortality rate into its natural and fishing mortality components by the method of Hearn *et al.* (1998).

Here, we use the twice-a-year tagging model as a base case and investigate four options for improving precision of natural and fishing mortality and tag-reporting rate estimates: increasing the number of animals tagged per tagging event per sex from 500 to 750, increasing the number of tagging events per year from two to three, increasing the tag-reporting rate from 0.22 to 0.5, and increasing the duration of the study by 3 and 5 additional years of once-a-year tagging.

Simulations are based on two tagging events per year, 500 lobsters tagged per sex per tagging event, 4 years of tagging, and 6 years of recaptures, except for increasing the duration of the study where there are 3 years of two tagging events per year followed by 3 or 5 years of one tagging event per year.

Precision and bias of estimated parameters were investigated with Monte Carlo simulation techniques. As all of the tagging models are product multinomial, the simulations are conveniently performed by the program SURVIV, described by White

(1983). The standard error and mean estimates were obtained with the PROC SIMULATE command.

Relative standard error (Rse), was obtained as

$$Rse = \frac{\overline{SE}}{\overline{X}}$$

where \overline{X} is the mean of the simulated estimates of a parameter and \overline{SE} is the mean standard error of the estimate (taken over all simulated data sets).

Unless otherwise specified, all simulations were conducted with simulated values for tag-reporting rate (λ) of 0.22, natural mortality (M) of 0.1 year^{-1} , and fishing mortality (F) of 1.0 year^{-1} . These values were chosen to be realistic for the rock lobster stock off north-western Tasmania. Frusher and Hoenig (in press) obtained estimates of fishing mortality in excess of 1.0 year^{-1} and reporting rate around 0.2. Their estimate of natural mortality was zero. For these simulations, tag-induced mortality and tag shedding were set to zero. Four years of tagging and six years of recaptures were simulated to parallel the tagging study conducted off north-western Tasmania. As with the model developed by Frusher and Hoenig (in press), tagging began in May of year 1 (*i.e.*, during the fishing season) and ended in May of year 4. Fishing and natural mortality were simulated on the basis of the amount of fishing effort and portion of year, respectively, that occurred between tagging events in the fishery from 1991 to 1997. Consequently, year 1 simulations were based only on the fishing effort undertaken between the May tagging and the close of the fishing season (17% of the total effort in the 1991–92 fishing year). Simulations were based on 500 lobsters tagged per sex per tagging event, as this number was achievable in the above-mentioned study. To determine the effect of additional years of tagging (*i.e.*, increasing duration of the study), we conducted additional simulations with six and eight years of tagging and six and eight years of recaptures, respectively. For each model, 1500 simulations were undertaken.

We also looked at the costs associated with various study designs. Costs of tagging were derived from costs incurred during tagging studies conducted from 1992 to 1995. These costs included vessel charter (A\$1800 per day), travel (A\$450 per trip)

and accommodation of a field officer (A\$100 per day), and costs of tags (A\$0.50 per tag). No rewards were offered for tags. Costs did not include the salary of the field officer, as it was assumed that the officer would be a regular employee of the research section. The opportunity costs incurred because the field officer might have undertaken other scientific activities have not been considered. Both the travel cost and the cost of tags are set for each trip irrespective of the number of days required to tag 500 lobsters of each sex. The shorter the trip the greater would be this contribution on a per-day basis, but both these costs are minor compared to the cost of vessel charter. The difference in fixed costs between a 5- and a 20-day trip is approximately 5% on a daily basis.

4.4 Results and Discussion

4.4.1 *Baseline simulations*

In the baseline simulations the most precise estimates of fishing mortality were obtained for years 2 to 5 (Table 4.2). The higher *Rse* of the first year's fishing-mortality estimate was expected given that the simulations mimicked the real tagging study and only 17% of the fishing effort in the first year was expended between tagging and the end of the fishing year. The higher *Rse* of the estimate for the last year is assumed to be due to the low number of tag returns. No new tags were available for capture after the May tagging event in year 4, and the high exploitation rate would leave few tags available for recapture from all previous tagging events. Under high exploitation, therefore, tagging might need to be maintained to yield low relative standard errors of fishing-mortality rate estimates. Although tag-reporting rate was simulated as low (0.22), the *Rse* of approximately 0.15 suggests that this parameter is estimated relatively precisely. In contrast, the *Rse* of the natural mortality estimate was high, indicating the poor precision in estimating this parameter from the twice-a-year tagging model.

Hearn et al (1998) found that they were able to estimate tag reporting rate and natural and fishing mortality rates reasonably well in simulated situations where F was only twice M and tag reporting rate was only as low as 50%. In contrast, the annual fishing-mortality estimates for the Tasmanian rock-lobster fishery (Frusher and Hoenig in press) are 10 times the estimate of M , and tag-reporting rate is less than half.

4.4.2 Increasing the number of lobsters tagged per tagging event

The most marked improvement in the relative standard errors of fishing and natural mortality estimates was obtained when the number of tagged lobsters was increased from 250 to 500 per sex per tagging event (Figure 4.1 *a* and *b*). At 500 tags per tagging event per sex, the relative standard error (*Rse*) of the natural mortality estimate was 1.03 (Table 4.2, base case).

The *Rse* of the natural mortality estimate improved by only 13% if fishing mortality was held constant over all years (Figure 4.1a), but the actual gain in precision depended on the degree to which the assumption of constant fishing mortality is met. Furthermore, Frusher and Hoenig (in press) outlined the advantages of estimating fishing mortality for each year. Even if 2000 tagged lobsters of each sex were released twice each year, the *Rse* of estimates of *M* only declined to 0.59, when *F* was estimated annually. The improvement was marginal, to 0.50, when *F* was held constant over all years, but the logistics of tagging so many lobsters each year are unrealistic in the Tasmanian fishery. Increasing the number of tags and using a model with fewer parameters may therefore not suffice for obtaining precise estimates of natural mortality rate.

The selection of 500 lobsters of each sex to be tagged at each survey appeared appropriate, as the greatest improvement in *Rse* of estimates of *F* and *M* was achieved when the numbers tagged increased from 250 to 500 per sex (Figure 4.1). Increasing the number of tagged lobsters each tagging event improved the *Rse* of annual fishing mortality rate estimates by approximately 18% when the number of lobsters tagged increased from 500 to 750 per sex (Table 4.2). Although these improvements may appear worthwhile, the *Rse* of fishing-mortality rate estimates were already relatively low, varying from 0.13 to 0.15 for years 2 to 5 (Table 4.2). Managers therefore need to consider the precision required for appropriate management decisions.

Tag-reporting rate showed marginally better improvements in *Rse* than did *F*'s as the number of lobsters tagged was increased (Figure 4.1 *a*). Whether *F*'s were calculated for all years combined or annually made no difference in the improvement of the *Rse* of tag-reporting rate. Like *F*'s, tag-reporting rate was relatively precise (*Rse* = 0.15) at 500 tags per sex per tagging event.

4.4.3 Increasing the number of tagging events per year

Natural mortality estimates showed a 15% improvement when frequency of tagging was increased from twice per year to three times per year (Table 4.2).

Increasing the frequency of tagging improved the precision of all but one of the fishing-mortality estimates by approximately 20% (Table 4.2). For the reasons mentioned above, the low, 5% improvement in Rse of the first year's F estimate is not surprising..

Tag-reporting rate showed the greatest improvement in Rse with increasing frequency of tagging, although as already mentioned, the Rse of tag-reporting rate is low.

Increasing either the number of lobsters tagged per event or the number of events per year involved tagging 1500 lobsters per sex per year. The additional tagging period per year produced only minor gains in precision compared to tagging more lobsters each tagging event.

4.4.4 Increasing the tag-reporting rate

Frusher and Hoenig (in press) attributed the poor precision in natural-mortality estimates to the high exploitation rates and low tag-reporting rate. As tag reporting rate is low, increasing it could improve estimates. Ways to do so include releasing tags with substantial rewards (Nichols *et al.* 1991; Pollock *et al.* 1995; Pollock *et al.* in press) and increasing fishers' awareness of the importance of returning tags. The latter could be achieved by means of regular visits to the fishing ports to speak with both vessel skippers and crews. Increasing tag-reporting rate from 0.22 to 0.5 improved the Rse of estimates of M by 35% (Table 4.2). A further increase in tag-reporting rate to 0.7 improved the Rse by an additional 24%. Significant improvements in tag-reporting rate are therefore required to improve the precision of estimates of M , although a 281% improvement in tag-reporting rate still resulted in relatively high Rse values of 0.5 for estimates of M .

Increasing tag-reporting rates from 0.22 to 0.5 and 0.7 decreased the Rse of F 's by approximately 40% and 55%, respectively (Table 4.2).

4.4.5 Increasing the duration of the study

To separate the total mortality rate into its natural and fishing mortality components, one must obtain an estimate of the tag-reporting rate. Several means of doing so, tagging more often (Hearn *et al.* 1998; Frusher and Hoenig in press) or using high-reward tags, planted tags, or a program of catch sampling (Pollock *et al.* 1995) all have associated costs. Once reporting rate has been estimated, it can be inserted as a fixed parameter into annual tagging models if the data analyst is confident that reporting rate is not likely to have changed appreciably over time. If ongoing estimates of fishing mortality are required, then annual tagging models could be applied for several years and interspersed with models with two tagging events per year to ensure that an appropriate tag-reporting rate is being used.

To indicate the precision of F and M estimates under the above scenario, we simulated a model in which 3 years of twice-yearly tagging were followed by 3 years of once-yearly tagging. Tagging events were based on the Tasmanian lobster fishery; tagging occurred before the fishing season for 6 years and after the female fishing season for the first 3 years only. The model was based on tagging of 500 lobsters of each sex per tagging event; reporting rate was simulated as 0.22, natural mortality as 0.1, and fishing mortality as 1.0. This model gave a 63.7% improvement in the Rse of estimates of M over those for the base case with 4 years of tagging twice a year and 6 years of recaptures (Table 4.2). The improvement in Rse of the estimate of M over that for the previous options is considered to be due to the three extra years in which tagging was undertaken.

Knowledge of the extent of improvement in precision of M estimates with increased duration of tagging is required for planning of tagging studies. To evaluate the potential gains in precision of estimates of M associated with increasing the duration of tagging, we undertook simulations with 1 to 5 additional years of tagging after an initial 3 years of twice-a-year tagging (Figure 4.2). Although the Rse value started to plateau after 2 years of once-a-year tagging, addition of a third year did improve the Rse of the M estimate by 13.7%. Improvements in Rse of less than 10% were achieved after the third year of once-a-year tagging. Achievement of an M estimate with a Rse of less than 0.3 would be unlikely if tag-reporting rate remained low and fishing mortality continued to be high, irrespective of the duration of the study.

All Rse values of F 's except for the last year were 0.1 or less (Table 4.3). Although still low, the higher Rse value of the last year was considered to result because it was the last year of tagging. We confirmed this conclusion by running the model with an additional two years of one-a-year single tagging (*i.e.* 3 years of tagging twice per year followed by 5 years of tagging once per year). In the 8-year model, the Rse of the final-year fishing mortality estimate was 0.15, but the Rse of fishing mortality for year 6 was 33% lower than that for the 6-year model (Figure 4.3).

As increasing the tag-reporting rate provided the greatest improvement in the Rse estimates of the previous three options, we investigated the effect of improving tag-reporting rate on the 6-year model (3 years of tagging twice per year followed by three years of tagging once per year). Improving the tag-reporting rate from 0.22 to 0.5 improved the Rse of annual F 's by 35 to 41% and M by 43% (Table 4.3). Thus a combination of prolonged tagging and improved participation by fishers appears the most promising way of improving the precision of estimates of F 's and M from a heavily exploited population.

An important assumption in using the combined model, in which twice-a-year tagging is followed by once-a-year tagging, is that reporting rate is constant. Tag-reporting rate might vary for a number of reasons, however, including a possible increase in fishers' willingness to cooperate when management plans for the fishery are favourable or a decrease when they are unfavourable. Fishers can also become 'bored' with continuing to return tags once the novelty of the system wears off. Because tag-reporting rate can change, we recommend that twice-a-year tagging be interspersed with once-a-year tagging if long periods of tagging are being considered.

The improvements in precision that resulted from increases in the tag-reporting rate demonstrate that model estimates are sensitive to such changes. Thus confidence that tag-reporting rate is not changing is important. Fortunately, the low Rse values of tag-reporting rate show that it can be estimated with good precision.

4.4.6 Cost analysis

We have demonstrated which options provide the best improvements in estimates of fishing and natural mortality, but the cost effectiveness of the various options must be considered because tagging can be expensive. Mean catch rates of male and female lobsters for each survey period were obtained from surveys undertaken from May

1992 to May 1995. The catch rates are averaged for each survey period over all years (Table 4.4). May has the lowest catch rate, particularly of females, and therefore requires the greatest number of days for tagging of any given number of female lobsters.

Increasing the number of tags involves tagging an extra 250 lobsters per sex during September and May and was the most expensive option (Table 4.5). Increasing the frequency of tagging involved tagging 500 additional lobsters per sex during February. For these two options, the total numbers of tagged animals released per year are the same. The lower cost associated with undertaking an extra survey reflects the higher catch rates in February than in May. Vessel-charter costs account for approximately 93% of the total costs, so the number of days required to tag lobsters determines the cost efficiencies. Including a February tagging survey involves an extra 16 days, whereas increasing the number of tags from 500 to 750 in May and September requires 21 extra days. The cheapest option was to add an additional tagging year to the study. It improved the Rse of the estimate of M and the penultimate year's F in addition to estimating F for an additional year.

The costs associated with the option of increasing tag-reporting rate are unknown, and the improved results cannot be guaranteed. An increased publicity campaign and the offer of rewards for return of tags might lead to substantially higher tag returns for a modest price. Nichols *et al.* (1991) found substantial improvements in tag-reporting rate with the use of reward tags. Such tags should be considered in the design of tagging projects, given the improvements in the precision in parameter estimation when tag-reporting rate increased.

Although the costs of tagging appear high, approximately 93% of the cost is associated with vessel charter. Using research vessels or forming alliances with fishers under which charter costs are reduced can minimize this cost.

4.5 Conclusions

This study has shown that M is a difficult parameter to estimate precisely from a fished stock where exploitation rate is very high and M and tag-reporting rates are low, even if 'state of the art' tagging models are employed. Increasing either the number of lobsters tagged or the number of tagging surveys per year had limited impact on improving the relative standard error of F_i or M estimates. Improving tag-

reporting rate improved the precision of F_i and M estimates substantially. An increase in the tag-reporting rate from 0.22 to 0.5 doubled the improvement in the precision of F_i and M estimates that result from an increase in either the number of lobsters tagged or the frequency of tagging. Unfortunately, improvements in tag reporting rate cannot be guaranteed, although reward tags should be considered. The best improvement in the precision of F_i and M estimates resulted when the duration of the study was increased. Twice-a-year tagging is required to estimate tag-reporting rate if no other options for estimating tag-reporting rate are available. Costs can be minimized if tagging is undertaken only once per year after an initial period of twice-per-year tagging. Under the assumption of constant reporting rate over time, a model based on once-a-year tagging after an initial period of twice-a-year tagging provided the lowest relative standard errors of F_i and M , and these can be further improved if tag-reporting rate can be increased. Increasing the duration of the study was also the cheapest option and had the additional benefit of providing estimates of F for additional years.

Table 4.1. Expected recoveries in a multiyear tagging study. N_i is the number of lobsters tagged in event i , f_i is the expected fraction of tags recovered in period i (between tagging events i and $i+1$), and S_i is the fraction of lobsters that survived in period i .

Tagging event	No. tagged	Expected recoveries in period			
		1	2	3	4
1	N_1	N_1f_1	$N_1S_1f_2$	$N_1S_1S_2f_3$	$N_1S_1S_2S_3f_4$
2	N_2	—	N_2f_2	$N_2S_2f_3$	$N_2S_2S_3f_4$
3	N_3	—	—	N_3f_3	$N_3S_3f_4$

Table 4.2. Improvements in the relative standard error (*Rse*) of annual fishing mortality (F_t), natural mortality (M), and tag-reporting rate (λ) estimates resulting from an increase in (A) number tagged per tagging event per sex from 500 to 750, (B) frequency of tagging from twice per year to three times per year, (C) tag-reporting rate from 0.22 to 0.5, (D) tag-reporting rate from 0.22 to 0.7, (E) the duration of tagging by 3 additional years of once-a-year tagging, and (F) the duration of tagging by 5 additional years of once-a-year tagging. Simulations are based on $M = 0.1$, $F = 1.0$, and $\lambda = 0.22$, two tagging events per year, 500 lobsters tagged per sex per tagging event, 4 years of tagging, and 6 years of recaptures, except for E and F where there are 3 years of two tagging events per year followed by 3 (E) or 5 (F) years of one tagging event per year.

Parameter	Base case	% improvement in <i>Rse</i> from base case					
	<i>Rse</i>	A	B	C	D	E	F
M	1.03	10.8	14.6	34.9	58.4	63.7	72.0
λ	0.15	19.4	24.6	—	—	53.8	67.8
F_1	0.27	18.1	4.5	34.9	46.3	63.6	64.3
F_2	0.15	18.8	20.5	40.9	55.2	44.1	46.1
F_3	0.14	18.9	21.0	41.2	55.7	40.5	43.7
F_4	0.13	18.9	21.7	40.8	55.1	39.8	45.7
F_5	0.14	18.4	21.3	39.5	53.5	29.1	42.9
F_6	0.25	16.9	21.4	38.3	52.5	44.1	62.5

Table 4.3. Relative standard errors (*Rse*) of fishing (*F*) and natural (*M*) mortality rates estimated from a model with 3 years of two tagging events per year followed by 3 years of a single tagging event per year. The percentage improvement in the *Rse* when reporting rate is increased from 0.22 to 0.5 is shown.

Parameter	<i>Rse</i> at 0.22	<i>Rse</i> at 0.5	% improvement in <i>Rse</i>
<i>M</i>	0.38	0.21	43
<i>F1</i>	0.10	0.06	35
<i>F2</i>	0.08	0.05	37
<i>F3</i>	0.08	0.05	39
<i>F4</i>	0.08	0.05	38
<i>F5</i>	0.10	0.06	41
<i>F6</i>	0.14	0.08	39

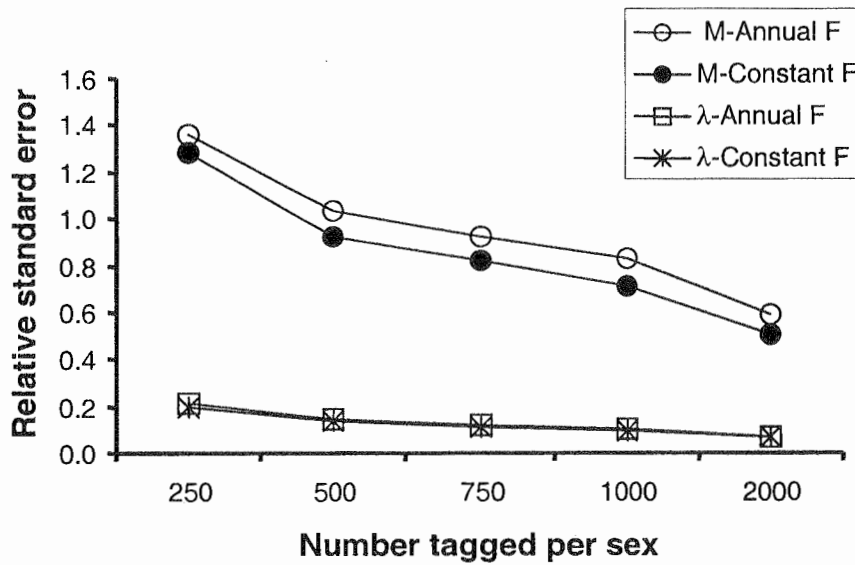
Table 4.4. Catch rate, estimated number of days, and associated costs required to tag 500 and 750 legal-size male (M) and female (F) lobsters for the three survey periods. Number of traps fished per day = 80. Daily costs based on the maximum days required to tag the specified number of lobsters for either sex (e.g. May = 33 days for 500 tagged lobsters and 49 days for 750 tagged lobsters).

Survey period	Catch rate (#/trap lift)		Estimated number of days to tag				Estimated daily cost (\$A)	
			500 lobsters		750 lobsters		500	750
	M	F	M	F	M	F	M	F
Feb.	0.44	0.40	15	16	22	24	1944	1934
May	0.30	0.19	21	33	31	49	1921	1917
September	0.53	0.61	12	11	18	16	1958	1946

Table 4.5. Comparison of costs associated with improving the precision of fishing, natural-mortality, and tag-reporting rate estimates.

Option	Cost A\$
Increasing number of tagged lobsters from 500 to 750 per sex for two tagging events in a year	\$42 050
Increasing frequency of tagging from two to three tagging events per year, each with 500 lobsters tagged per sex. The additional tagging event occurs in February.	\$31 100
Cost of each additional year of once-a-year tagging of 500 lobsters per sex. Tagging undertaken in September.	\$23 500

(a)



(b)

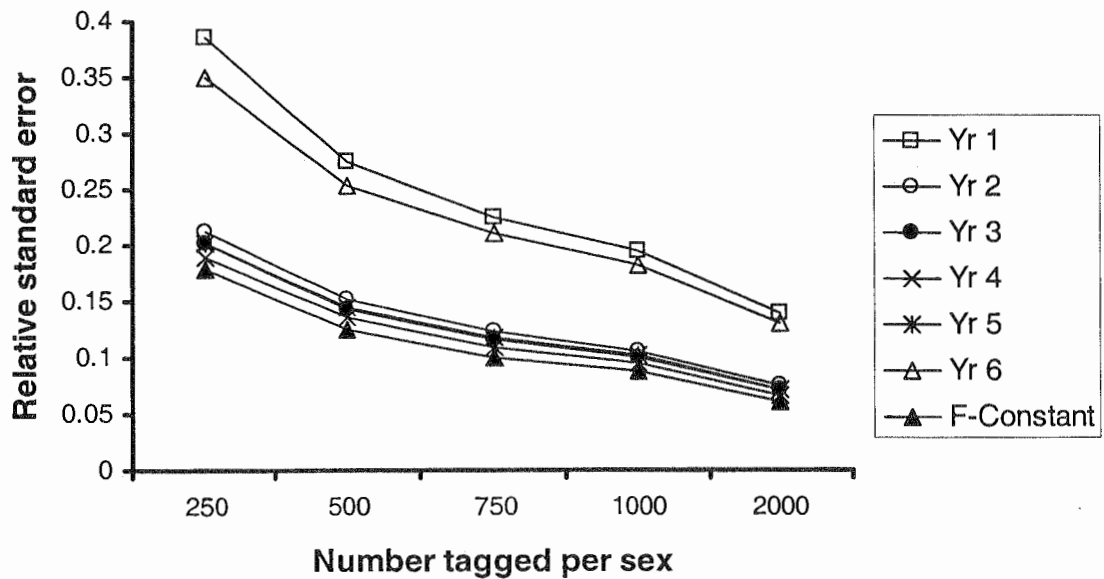


Figure 4.1. Improvement in relative standard error of (a) natural mortality (M) and tag-reporting rate (λ) estimates and (b) fishing mortality (F) estimates resulting from increased number of lobsters tagged. Values are presented for cases in which fishing mortality was estimated annually or set to be constant over all years.

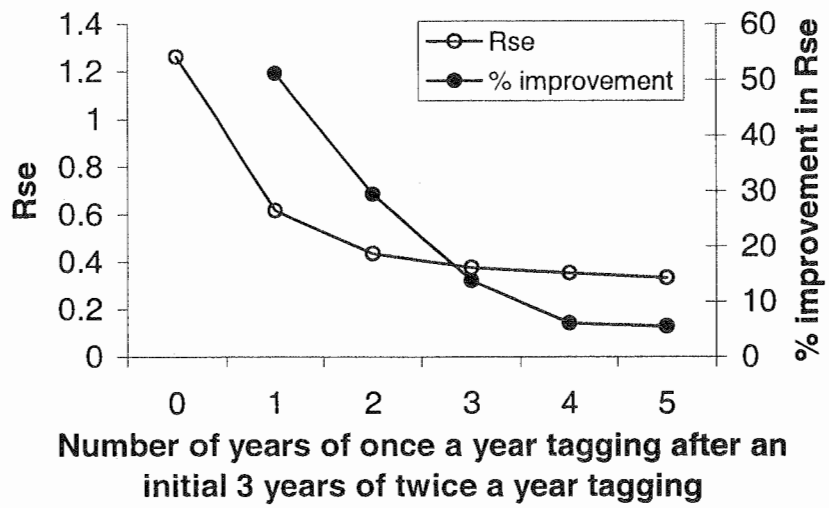


Figure 4.2. Comparison of *Rse* of natural-mortality estimates as the number of years of once-a-year tagging increase after an initial 3 years of twice-a-year tagging.

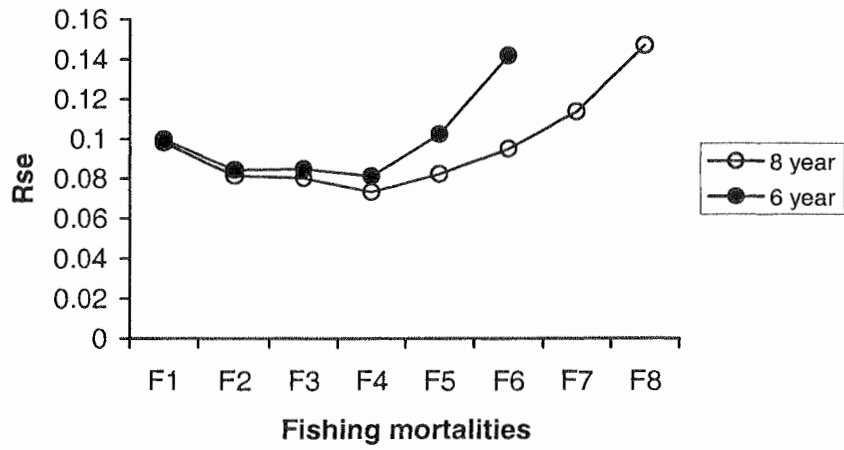


Figure 4.3. Comparison of the relative standard error (Rse) of fishing-mortality estimates from models with 3 years of twice-a-year tagging followed by either 3 years (6-year) or 5 years (8-year) of once-a-year tagging.

*5. Impact of lobster size on selectivity of traps for southern rock lobster (*Jasus edwardsii*).*

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Science)

5.1 Abstract

Most lobster fisheries are characterized by high exploitation rates. This has led to substantial declines in the size structures of the populations over time as larger lobsters have been removed. Although both scientists and fishers have suggested that size related hierarchies could impact on lobsters entering traps, the effect of the size change on the selectivity of lobster traps as a populations size structure changes has not been investigated. This paper demonstrates that large lobsters affect the entrapment of small lobsters and that this behaviour affects the selectivity of lobster traps. Both spatial and temporal (within season) factors were found to affect the selectivity curves. Spatial differences in selectivity were attributed to the broader size range of larger lobsters found in regions of faster growth. Temporal differences were attributed to the decline in larger lobsters over the course of a season due to exploitation. There are also differences in trap selectivity between the sexes.

5.2 Introduction

Selectivity is a measure of the relative catchability of one component of a population (e.g., size class) to other components of the population for those animals that encounter a given type of fishing gear. Length-based estimates of fishing gear selectivity are crucial in fisheries science if population parameters are to be estimated from length-structured catch data. Although selectivity of a variety of fishing gears has been estimated, recent papers have demonstrated that selectivity for a particular gear can change both spatially (Anganuzzi et al. 1994; Addison and Lovewell 1991) and temporally (Myers and Hoenig 1997).

Over the last 30 years the Tasmanian rock lobster fishery (*Jasus edwardsii*) has seen a substantial decline in the biomass of legal-sized animals to a level of approximately 6 percent of an unharvested population (Frusher 1997). This has resulted in substantial changes to the size structure of the population with the bulk of the legal sized biomass being recently recruited lobsters. Larger lobsters are becoming increasingly rarer in the catch (Frusher 1997).

In addition to having a legal sized biomass based primarily on recruits, the Tasmanian rock lobster fishery has high exploitation rates in most regions (Frusher et al. 1998; Frusher and Hoenig unpublished data). Within a fishing season there is a substantial shift in the size structure of the population from large (recruited) lobsters to small (undersized) lobsters as large lobsters are harvested.

Most comparisons of the size composition of a population over both spatial and temporal scales assume that selectivity is constant and that the size distributions can be compared directly. Assessment techniques which compare size classes such as the change-in-ratio technique (Pollock and Hoenig 1998; Frusher et al. 1997, 1998) also assume that the selectivity of the size classes being considered does not change between surveys.

Although it is recognized that behavioral changes such as those associated with molting affect catchability of lobsters (Morgan 1974; Newman and Pollock 1974; Krouse 1989), behaviour also affects selectivity as Frusher et al. (1998) found larger lobsters to molt earlier during their molting period. Another factor that could affect selectivity may be the presence of size-related dominance hierarchies. Olsen (1958) and Winstanley (1977) suggest that dominant larger sized lobsters could prevent smaller lobsters from entering traps. Addison (1995) and Miller (1979, 1990) suggest that fishing pressure would remove these dominant lobsters and thus influence the size distribution of lobsters entering a trap during the fishing season.

Larger lobsters have been found to have higher catchabilities than smaller lobsters and males to be more 'catchable' than females (Miller 1989). Different styles of traps and different areas have also been found to produce different trap selectivity (Addison and Lovewell 1991). However, there have been no studies on the impact on trap selectivity associated with harvest of a population over time.

This paper uses tagging data collected from fishery independent catch sampling

surveys undertaken from 1992 to 1998 to evaluate temporal and spatial changes in selectivity of rock lobsters in research traps around Tasmania. Tagging data were also obtained from a study of an unfished population in a marine reserve.

5.3 Materials and methods

5.3.1 Reserve sampling

5.3.1.1 Site, procedure and sampling gear

The Crayfish Point Reserve was established in 1971 and is located in the Derwent estuary in Tasmania (Figure 5.1). The reserve extends 800 m from the shoreline and covers an area of 1.0 square kilometer. Habitat maps of the reserve indicate that approximately 40% of the reserve is lobster habitat. In 1999, two surveys were undertaken from 28th January to 18th February and from 30th March to 10th April. During each survey 50 baited lobster research traps were set each day and hauled approximately 24 h later during the following morning. Traps were set randomly on the lobster habitat and each region of lobster habitat was sampled at least once during each survey. All lobsters caught in each trap were measured to the nearest mm carapace length (CL) and sexed. All lobsters were transported to an adjacent shore based holding facility at the Tasmanian Aquaculture and Fisheries Institute's Marine Laboratories. Lobsters greater than 80 mm CL were tagged ventrally in the flesh of the first abdominal segment (Hallprint T-bar anchor tag; TBA1, Hallprint Pty. Ltd., 27 Jacobsen Crescent, Holden Hill, South Australia 5088, Australia). All lobsters were released back into the reserve at the completion of each survey.

Research traps were similar to the square steel traps that are becoming increasingly popular in southern parts of the fishery and are described in Frusher et al. (1998). The main difference between research traps and commercial traps was that research traps had no escape gaps, which are mandatory for commercial fishing operations. The escape gaps are to allow undersized lobsters to escape without having to be handled during the fishing operation.

5.3.1.2 Investigating the potential interaction between different sizes in traps

To investigate whether small lobster catchability was influenced by the presence of larger lobsters, three analyses were undertaken for the reserve data: (i) Catch rates (number of lobsters per trap lift) of small (< 90 mm CL) and large lobsters (> 140 mm

CL) were compared for each day of both surveys. (ii) Correlations between the number of small (< 90 mm CL) and large (> 109 mm CL) lobsters caught in each trap of both surveys were undertaken. Large lobsters were defined as being greater than 109 mm CL for this analysis so that the results could be compared with the regional fishing ground data. Southern regions had no lobsters in the greater than 140 mm CL category. (iii) Selectivity curves based on recaptures of tagged lobsters caught during the first four days and last four days of the second survey were compared.

There were no other reserve areas available to determine if there are any changes in catch rates when lobsters are not removed. However, lobster sampling was identical in surveys undertaken in the reserve in January and December 2000 with the exception that lobsters were immediately released after being measured, sexed and tagged. We use these surveys to check on the possibility of temporal changes in catch rate occurring independently of removals.

Because the data are not bivariate normally distributed, we used Spearman's rank correlation coefficient to test for significance of association.

5.3.1.3 *Estimating selectivity*

Recaptures in the second survey of lobsters tagged in the first survey were used to estimate selectivity. Selectivity could only be determined for male lobsters as female lobsters began to molt between surveys (P. Zeigler Tasmania Aquaculture and Fisheries Institute, University of Tasmania, GPO Box 252-49, Hobart, Tasmania, Australia 7001.).

Selectivity (ϕ_l) was determined for each 10 mm CL size class by the proportion of tags returned in the second survey from tagging in the first survey.

$$(1) \quad \phi_l = \frac{(N_{Rl} / N_{Tl})}{\max_l (N_{Rl} / N_{Tl})}$$

where N_{Tl} is the number of lobsters tagged in size class l during the first survey and

N_{Rl} is the number of lobsters of size class l tagged in survey 1 which were recaptured in survey 2. The denominator serves to standardize the selectivity estimates to the interval $[0,1]$. Selectivity was not calculated for size classes where less than 20 lobsters were tagged in the first survey.

5.3.2 Research sampling on the fishing grounds

5.3.2.1 Sites, procedure and sampling gear

Three regions in Tasmania were used to investigate the interaction between different sizes in lobsters caught in traps (Figure 5.1). These regions span a range of growth rates from slow growth in the southern region to fast growth in the northern region (Frusher 1997). Only the southern and eastern regions that were sampled from 1992 to 1998 in a fishery independent catch sampling project were used to determine selectivity in fished regions (Figure 5.1). Selectivity could not be determined in the northwestern region because sites were not re-sampled during each survey and thus there were no recaptures suitable for determining selectivity.

The Tasmanian fishing season opens in November and closes at the end of April and August in the following year for female and male lobsters, respectively. Surveys were undertaken three times each year. A pre-season survey was undertaken in October/November, a second survey was conducted in March prior to the female molt (April to May) and the final survey was conducted in July/August prior to the male molt (September to October) (Frusher et al. 1998). On the south and east coasts there are 4 sampling sites and, if weather allowed, we attempted to sample each site twice during each survey. Traps are set randomly within each site, although the size of the sites is relatively small ($< 0.5 \text{ km}^2$) and thus the majority of lobster habitat is sampled each survey. Each survey consisted of eight and nine overnight fishing shots in the eastern and southern regions respectively. Each shot involved the setting of 50 identical research lobster traps. Tagging was conducted during each survey. To ensure that the collected data represented the catch from the commercial fishery, sites were selected after consultation with fishers.

Sampling gear was the same as that used in the reserve. From each trap, all lobsters were measured and sexed and in regions where catch rates of lobsters were high, tagging was restricted to approximately 1000 lobsters per survey. This was achieved by tagging the first 200 lobsters per shot. All lobsters (newly tagged, recaptures and

untagged) were released back in their site of capture. Tagging surveys were used to determine selectivity when the difference in the mean of tagged and recaptured lobsters differed by less than 5 mm CL and 4 mm CL on the east and south coasts respectively.

5.3.2.2 *Investigating the potential interaction between different sizes in traps*

To investigate whether there was an interaction between the catchability of small and large lobsters the correlation was estimated for the number of small lobsters (< 90 mm CL) and large lobsters (> 109 mm CL) caught in individual research traps. As for the reserve data, Spearman's rank correlation coefficient was used to test for significance of association.

5.3.2.3 *Estimating selectivity*

On the commercial fishing grounds selectivity estimates could only be made from surveys where recaptures were obtained from lobsters tagged during each specific survey (e.g. only recaptures obtained during the March 1994 survey from lobsters tagged during the March 1994 survey could be used). Both molting and harvesting of legal sized lobsters affected the size distribution of tagged lobsters between surveys.

Selectivity (ϕ_{li}) was determined for each 5 mm CL size class by the proportion of tags returned on subsequent days after tagging during each survey.

$$(2) \quad \phi_{li} = \frac{(N_{Rli} / N_{Tli})}{\max_{li} (N_{Rli} / N_{Tli})}$$

where N_{Tli} is the number of tagged lobsters for each size class l during survey i and is derived from:

$$(3) \quad N_{Tli} = \sum_{j=1}^{J-1} T_{lj}$$

where T_{lj} is the number of lobsters of length l tagged on day j of survey i and J is the number of days in the survey, N_{Rli} is the number of recaptured lobsters for each size class l during survey i and is represented by

$$(4) \quad N_{Rli} = \sum_{j=2}^J R_{lj}$$

where R_{lj} is the number of lobsters of length l recaptured on day j of survey i .

Selectivity was not estimated for size classes where less than 20 lobsters were tagged. Multiple recaptures (the same lobster being caught more than once before the end of a sampling trip) were few and classified as a single recapture to minimise ‘trap happy’ behaviour.

The principal aims of the tagging component of the fishery independent study were to obtain growth and movement information and molt frequency to aid in elucidating exploitation rate estimates in the fishery. Unfortunately there was insufficient data to evaluate seasonal selectivity for each year and thus it was necessary to combine all tagging events from the same survey time period (e.g., March) over all years.

On several occasions tagging focused on specific size groups. To estimate selectivity, tagging needs to be random or needs to target the same size classes over all surveys to be summed. To determine if a tagging event could be used in the selectivity study, the mean length of all lobsters caught was compared to the mean length of all lobsters tagged. Selectivity was determined for each sex separately.

5.4 Results and Discussion

5.4.1 Interaction between different sizes in traps

5.4.1.1 Reserve

A significant negative correlation was found between catches of large and small lobsters in the reserve (Table 5.1). The correlations indicate that there is a sex effect as large females have a higher negative correlation with both small females and small males than large males have with either small females or small males. This is despite males reaching larger sizes than females.

To further explore the impact of large lobsters on catchability of small lobsters the daily catch rates of small and large lobsters during the reserve surveys were compared (Figure 5.2 a & b). Large lobsters showed a decline in catch rates as both surveys progressed while small lobsters showed a gradual increase before beginning to decline. This suggests that the presence of large lobsters in traps at the beginning of the survey was preventing small lobsters from entering the traps. As the number of large lobsters decreased due to removal from the reserve, the catch rates of small lobsters increased until their numbers started to decline due to depletion from sampling activities.

To determine if the patterns over time of catch by size group might be due to temporal processes occurring independently of removals, we examined the catch composition over time when animals were trapped but immediately returned to the water (Figure 5.3). The composition of the catch remained constant over time. This suggests that it is the reduced abundance of large lobsters that causes the catch rate of small lobsters to increase.

5.4.1.2 Fishing grounds

A significant negative correlation was also found between the number of small and large lobsters (regardless of sex) caught in traps during surveys of the Tasmanian rock lobster fishing grounds in each of the three regions surveyed (Table 5.1). With the exception of the correlation between large males and small females on the south coast, all negative correlations increase from southern to northern Tasmania as does the size range of lobsters caught (Figure 5.4). The reason for the high negative correlation for large males and small females in the south is unknown. The trend in the other

correlations would suggest a size related dominance hierarchy. Agonistic interactions between different sized *Homarus americanus* have been reported by Schrivener (1971), Miller (1995) and Karnofsky and Price (1980).

Similar to the situation in the reserve, large females from eastern Tasmania had a higher negative correlation with small lobsters of either sex than large males had with small lobsters of either sex. Both the correlations from the three regions of Tasmania and the catch rates obtained from the reserve suggest that the presence of larger lobsters affects the entrapment of smaller lobsters.

5.4.2 Selectivity

To use tag recapture information to determine selectivity we assume that the number of tagged lobsters within each length class in the survey area at the time of recapture is directly proportional to the number of tagged lobsters within the same size class that were released during the first survey. Size dependent effects such as molting, emigration, natural mortality, tag loss or tag induced mortality would bias selectivity estimates. Moulting between surveys was only found to occur for females in the reserve and female selectivity curves for these data have not been attempted. Pearn (1994) found virtually no migrations of lobsters in southern and eastern Tasmania, Treble (1996) found tag loss to be around 1% for *Jasus edwardsii* in southern Australia and Kennedy (1992) estimated natural mortality to be 0.1yr^{-1} . Frusher and Hoenig (unpublished data) found a combination of tag loss, tag induced mortality and tag reporting rate to be 0.23yr^{-1} in northwestern Tasmania. The magnitudes of these effects appear to be low over the study period. Thus, it does not appear likely that these effects can vary greatly by size class.

5.4.2.1 Reserve

The male selectivity curve for recaptures obtained in survey 2 from lobsters tagged in survey 1 shows a linear increase with size up to 170 mm CL (Figure 5.5a). To determine the effect of a size interaction on selectivity, selectivity was estimated for the first and last four days of the second survey period (Figure 5.5b). The start of the survey showed a gradual increase in the selectivity curve up to 145 mm CL which was followed by a sharp rise to a peak selectivity of very large (170 - 180 mm CL) lobsters. In contrast, the selectivity curve for the latter four days of the survey had

increased selectivity of smaller lobsters (< 120 mm CL) and a peak selectivity in the 140 - 170 mm CL range.

5.4.2.2 *Fishing grounds*

Male lobsters caught during surveys on the east coast of Tasmania show an increase in selectivity with size, peaking at the size grouping 5 mm CL above the legal size limit of 110 mm CL. (Figure 5.6).

The slope of the selectivity curve for the start of the fishing season is less than the middle and end of season curves. At the middle and end of the fishing season there is an increased selectivity for lobsters in the size range from 80 mm CL to 110 mm CL relative to the selectivity at the beginning of the season. The similarity between the middle and end of season selectivity curves would be expected as over 80% of the commercial catch on the east coast has been taken by the middle survey period (Frusher et al. 1998). This is reflected in the size structure of the population that shows a decline in the legal sized lobsters between the start and middle season and limited change between the middle and end of season (Figure 5.7a).

For female lobsters on the east coast, the difference in selectivity curves within a season is less pronounced (Figure 5.7b). The start and end of season selectivity curves both peak at the 90 - 100 mm CL interval and are similar in shape except for the 70 - 80 mm CL interval. The small number of lobsters tagged in this size interval possibly accounts for this discrepancy. The similarity between the curves at the start and end of the fishing season is expected as the commercial season for female lobsters closes after the middle season survey. After this survey, female lobsters molt and thus all legal sized lobsters for the forthcoming seasons are available for capture during the end of season survey. Although the size structure of legal sized female lobsters at the start and end of the fishing season is not identical (Figure 5.7 b), they are less different in shape compared to the middle-season size structure.

The analysis for the south coast is restricted to males as females represent less than 5% of the catch (Frusher 1997). There is very little difference between the selectivity curves for the three survey periods (Figure 5.7 c). Because of the slow growth and high exploitation in this region there are very few large lobsters. Less than 13% of the male catch comprised lobsters greater than 140 mm CL on the south coast whereas over a third of male lobsters were above this size on the east coast.

5.4.3 Comparison between sexes

Selectivity of male and female lobsters was compared for the east coast where there is a high probability of recapturing tagged lobsters of both sexes. To demonstrate the relative differences, the maximum fraction recovered from either sex was used to standardise both plots. In all survey periods, large males had the highest selectivity (Figure 5.8). Selectivity for both sexes was similar for lengths up to approximately the 90 - 100 mm CL interval (start and end) and the 80 - 90 mm CL interval (middle). Above these intervals female selectivity declined while male selectivity continued to increase. This may be related to maturity of females as the size at which 50% of females are mature on the east coast is just below the 80 - 90 mm CL interval (Frusher 1997). These results are similar to those reported by Miller (1995) who found the catchabilities of small male and female *H. americanus* to be similar but then to increase more rapidly for males.

5.4.4 Comparison between regions

Regional comparisons of selectivity can be made between males on the south and east coasts. In all periods within the fishing season, south coast and east coast male selectivity peaked at the size range below and above the legal size limit of 110 mm CL respectively (Figure 5.9). At the start of the fishing season undersized males above 80 mm CL from the South have a higher selectivity than east coast males. As the season progresses, the trends in the selectivity curves become similar.

The correlation data (Table 5.1) suggested that large males (> 109 mm CL) had a larger impact on small males on the east coast compared to the south coast. The regional difference between the selectivity curves at the start of the season is considered to be due to the large males impacting on small males and thus shifting the selectivity curve towards large lobsters. The similarity between the selectivity curves at the end of the season is expected, as the majority of legal sized lobsters (≥ 110 mm CL) would be removed by exploitation from both regions.

In conclusion, the above results support fishers' beliefs that large lobsters exclude small lobsters from traps. The correlation data demonstrated that if a large lobster is in a trap it is unlikely that there will be many small lobsters and vice versa. This negative correlation was strongest in regions where large lobsters were abundant. The catch data from the reserve demonstrated that when large lobsters were present, they

dominated the catches. As these large lobsters were removed, smaller lobsters became increasingly catchable. This behavioral interaction between small and large lobsters affects the selectivity of the traps. In regions where large lobsters are found, such as the east coast and the reserve in Tasmania, seasonal selectivity curves shift in favour of smaller lobsters as large lobsters are caught. In contrast, the selectivity curves for the southern Tasmanian region showed no differences between the start, middle and end of season. This was supported by the very low correlation between large and small lobsters and is assumed to be due to the lack of large lobsters in this region. The lack of large lobsters is due to the high exploitation rates (Frusher et al. 1998) and the slow growth (Punt et al. 1997).

The slow growth in the south suggests that it is physical size rather than age that is causing the hierarchical dominance. Due to their slower growth, lobsters around the minimum legal size limit of 110 mm CL would be substantially older in southern regions than in more northerly regions. The declining growth rate from northern to southern regions of Tasmania would also increase the age difference between small (< 90 mm CL) and large (> 109 mm CL) lobsters used in the correlation. This further supports the suggestion that it is size rather than an age related dominance hierarchy.

Although various authors (see Miller 1989, 1995) have found catchability of larger lobsters to be greater than smaller lobsters and that large males have higher catchability than larger females, there has been no exploration of the impact of harvesting on selectivity. This paper demonstrates that selectivity can change with harvesting, even over small time frames such as within a fishing season. Assessment models for lobster fisheries, which use fixed selectivity curves, are likely to overestimate recruitment and this would be compounded over time as legal sized biomass is fished down. Conversely, these models are likely to under represent recruitment in fisheries, such as the Tasmanian rock lobster fishery, where management controls are aimed at rebuilding legal sized biomass. We believe that the impact of varying selectivity on assessment of commercial lobster resources requires further investigation.

Table 5.1. Correlation results between small (< 90 mm carapace length (CL) (S) and large (> 109 mm CL) (L) male (M) and female (F) lobsters caught in the south, east and northern regions of the fishery from 1992 to 1998 and in a reserve in southeastern Tasmania in 1999. (Figure 5.1). Spearman's rank correlation coefficients are presented followed by their P-values. N is the number of pairs used in each correlation.

Region		LM:SM	LM:SF	LF:SM	LF:SF
South					
	Correlation	-0.1227	-0.3426	-0.0885	-0.0379
	P-value	0	0	0	0.0165
	N	4903	5179	4178	3997
East					
	Correlation	-0.3767	-0.2596	-0.4560	-0.3489
	P-value	0	0.0001	0	0
	N	3111	3496	2561	3083
North					
	Correlation	-0.5265	-0.5290	-0.6621	-0.6586
	P-value	0	0	0	0
	N	4904	5180	4179	3998
Reserve					
	Correlation	-0.2850	-0.3211	-0.3983	-0.4065
	P-value	0	0	0	0
	N	923	944	588	607

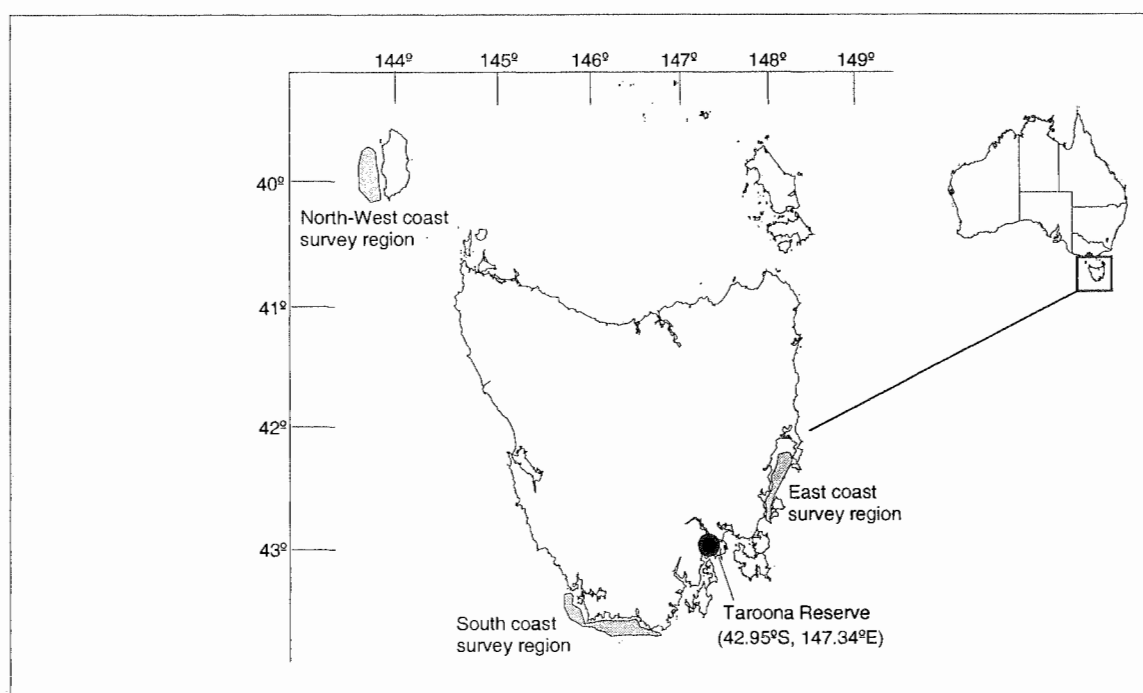


Figure 5.1. Map of Tasmania, Australia, showing the southern, eastern and northwestern regions surveyed and the location of the Crayfish Point (*Taroona*) Reserve.

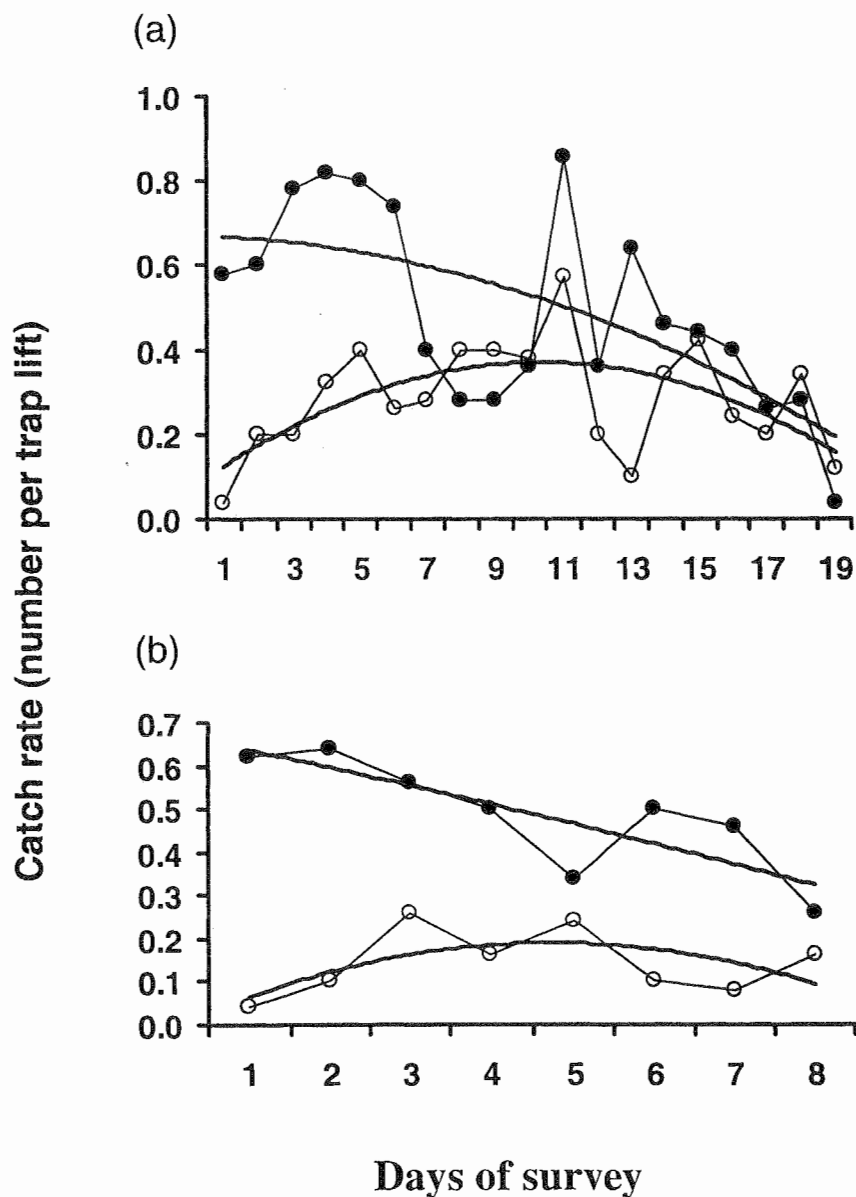


Figure 5.2. Comparison of catch rates in the reserve for small (< 90 mm carapace length (CL), open circles) and large (> 140 mm CL, closed circles) lobsters in a) survey 1 and b) survey 2. The polynomial equations describe the catch rate (CR) as a function of the day (t) of the survey. For survey 1 the equations are $CR = -0.0012t^2 - 0.0019t + 0.6716$ ($R^2 = 0.4303$) for large lobsters and $CR = -0.0028t^2 - 0.058t + 0.0701$ ($R^2 = 0.3531$) for small lobsters. For survey 2 the equations are $CR = -0.0007t^2 - 0.0383t + 0.6757$ ($R^2 = 0.6985$) for large lobsters and $CR = -0.0089t^2 - 0.0844t + 0.0096$ ($R^2 = 0.3357$).

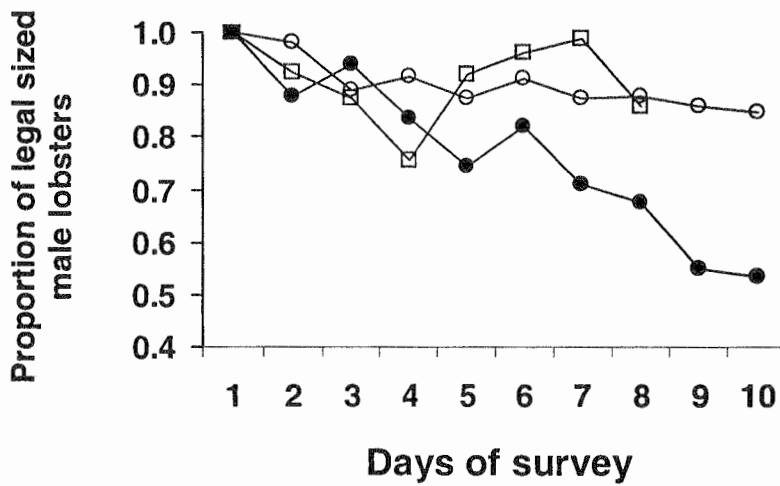


Figure 5.3. Proportion of legal size male lobsters caught during each day of three surveys undertaken in January 1999 (closed circles) and 2000 (open circles) and December 2000 (open squares). In January 1999 lobsters were removed from the survey site during the period of the survey. In January and December 2000, lobsters were released into the survey site immediately after capture.

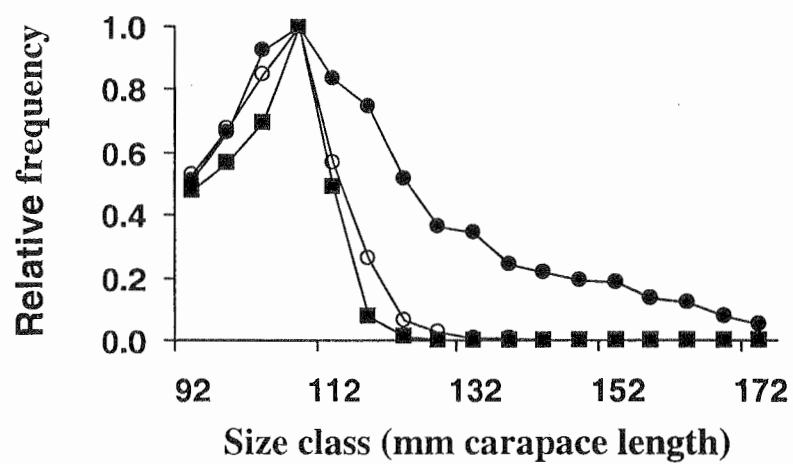


Figure 5.4. Regional size structure of male lobsters from north (N, closed circle), east (E, open circle) and south (S, closed squares) Tasmania.

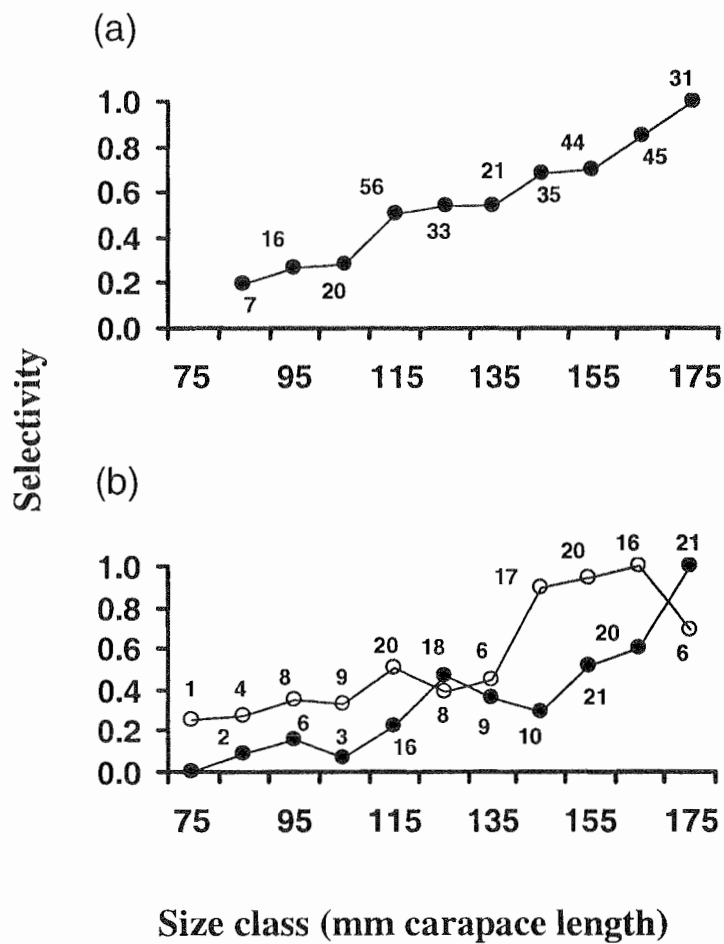


Figure 5.5. Estimated selectivity for male lobsters in the reserve for (a) the entire second survey and (b) the first five days (closed circle) and last five days (open circle) of the second survey. Numbers refer to numbers of recaptures.

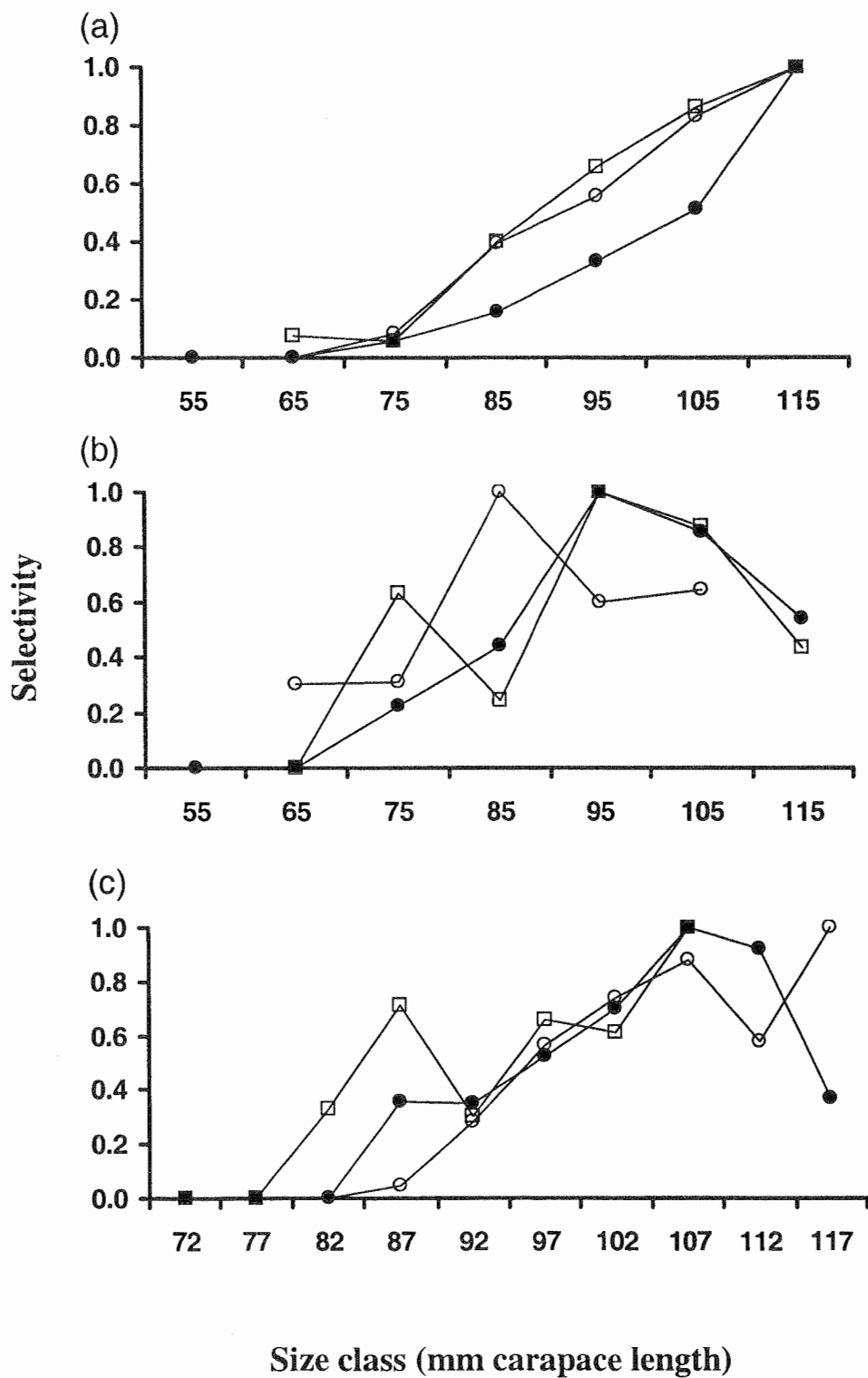


Figure 5.6. Estimated selectivity for (a) male lobsters from the east coast of Tasmania, (b) female lobsters from the east coast of Tasmania and (c) male lobsters from the south coast of Tasmania at the start (closed circle), middle (open circle) and end (open square) of the fishing season.

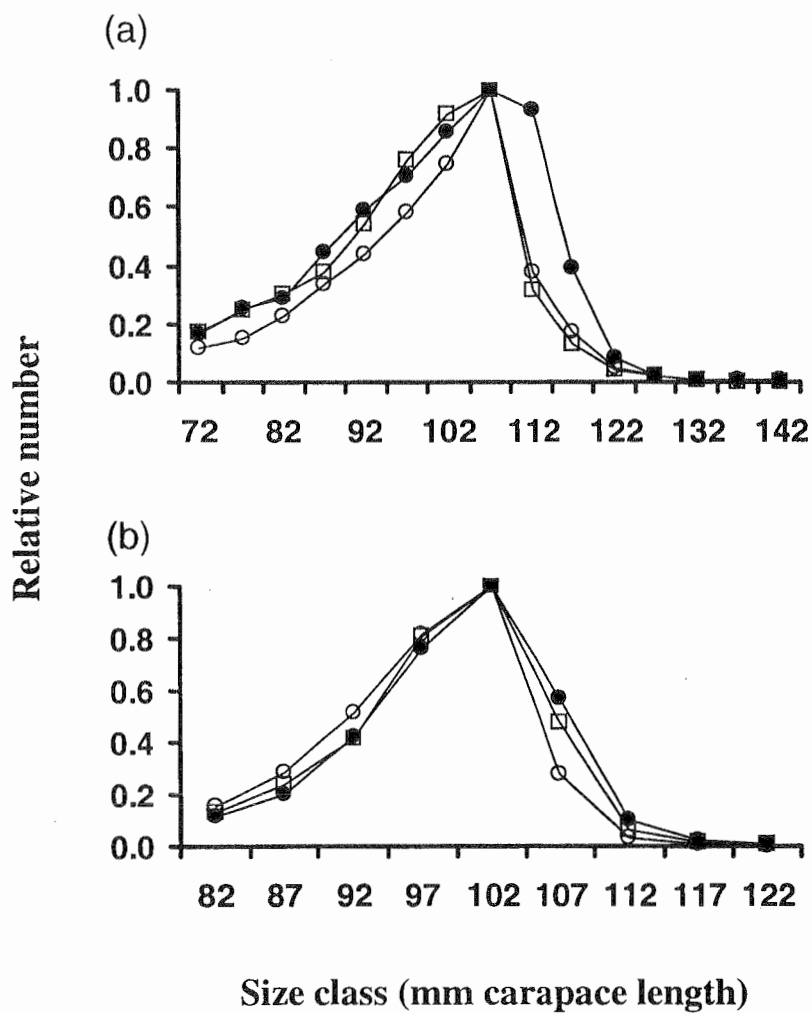


Figure 5.7. Relative size structure of (a) male lobsters caught in traps on the east coast, and (b) female lobsters caught in traps on the east coast at the start (closed circle), middle (open circle) and end (open square) of the fishing season.

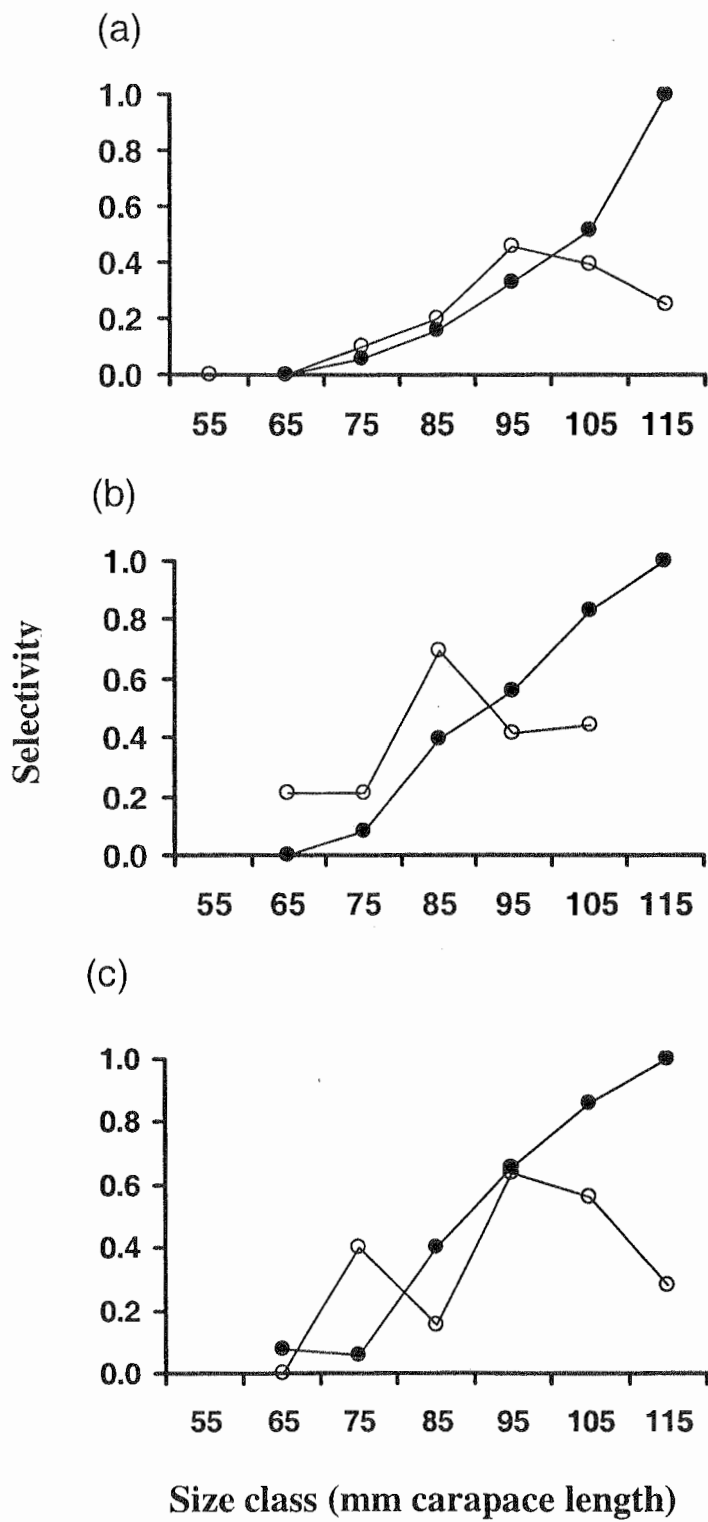


Figure 5.8. Comparison of selectivity for male (closed circle) and female (open circle) lobsters on the east coast for the (a) start, (b) middle and (c) end of the fishing season.

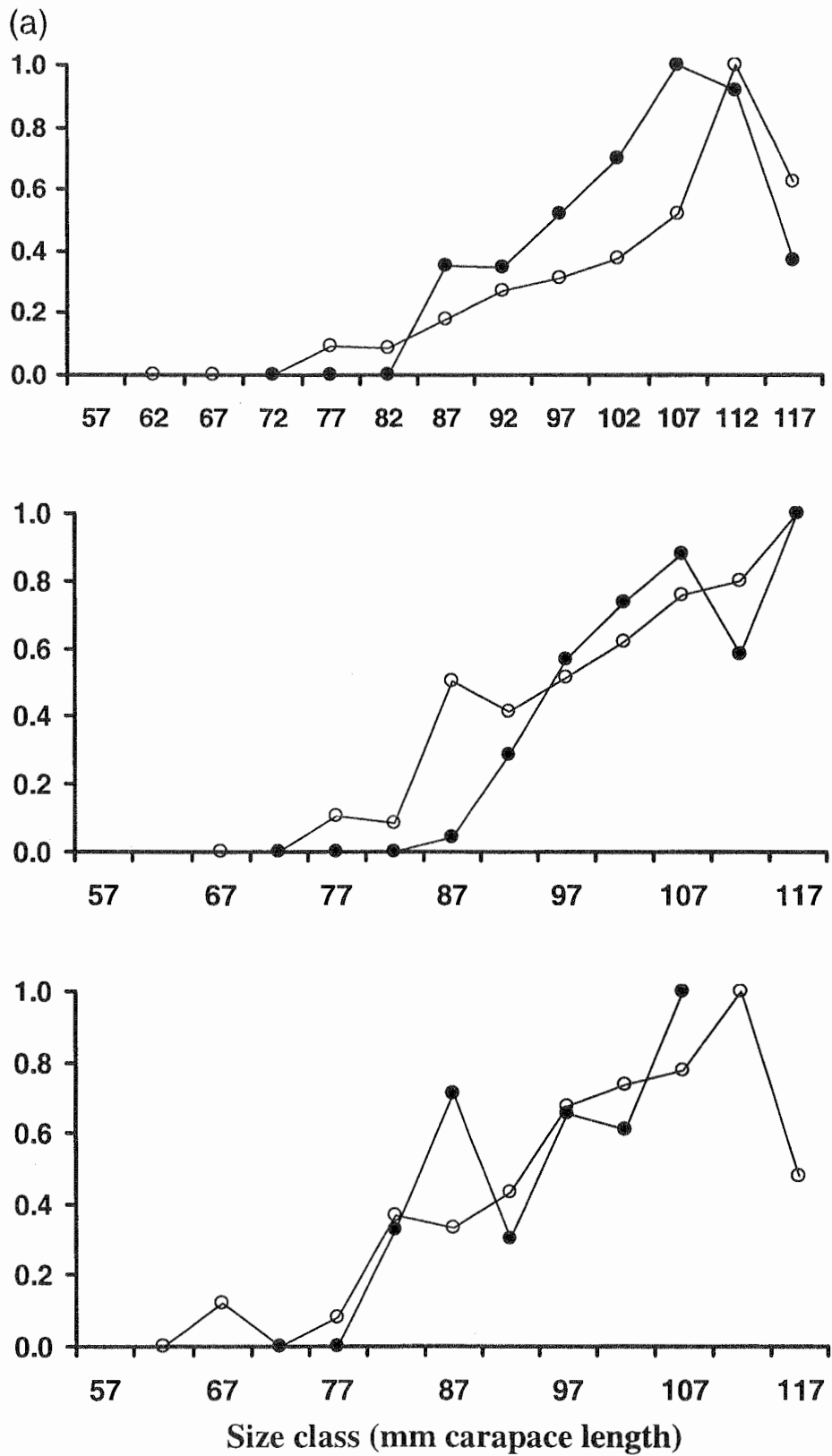


Figure 5.9 Comparison of selectivity for male lobsters between the south (closed circles) and east (open circles) coasts for the (a) start, (b) middle and (c) end of the fishing season.

6. Have changes in selectivity masked recruitment declines in crustacean trap fisheries?

6.1 Abstract

Recent developments in fishery assessment models are focused on management strategy evaluation. In these models, future trends in specified fishery parameters, such as biomass, are evaluated against a range of different harvest strategies (quotas, closed seasons, effort reductions etc.). Often, one or more parameter values used in the models are based on single studies of short duration. Consequently, there is no basis for providing for temporal variability in parameters. For example, in the Tasmanian rock lobster assessment model, selectivity is treated as a fixed effect. The actual size composition in the population is estimated by dividing the catch in each size class by the size-specific selectivity. Recently, trap selectivity has been found to vary with the size composition of the population being sampled. Larger lobsters were found to inhibit smaller lobsters from entering traps. We suggest that as larger lobsters are removed from the population by harvesting, smaller lobsters increase in the catch because they become more catchable. To determine the effect of a change in selectivity pattern on our perception of population composition, we applied selectivity curves that accounted for effects of size composition to sampling data from a population of lobsters that had seen a substantial decline in large legal sized lobsters by harvesting over the last 35 years. The results suggest that recent recruitment is lower than the recruitment that occurred in the 1960s, but this is masked in the unadjusted sampling data by changes in selectivity. This could contribute to an apparent lack of a stock-recruitment relationship.

6.2 Introduction

Lobsters constitute major fisheries around the world and have consequently received substantial attention from managers and scientists. This has resulted in considerable literature on lobster assessment (see Breen (1994), Addison (1997) and Hilborn (1997) for reviews). Over the last decade most of the advances in rock lobster

assessment have involved the development of new population models (Bergh and Johnstone, 1992; Walters et al., 1993; Punt and Kennedy, 1997; Walters et al, 1997). In summarising an international workshop on models used for assessing lobster stocks around the world, Hilborn (1997) found the general trend to be towards dynamic models fitted to observed length distributions. In Tasmania, Southern Australia, a spatially explicit, size-structured model (Punt and Kennedy, 1997) is used to undertake risk assessments of future harvest strategies for southern rock lobster (*Jasus edwardsii*) (Frusher, 1997; Frusher and Gardner, 1999).

Despite the advances in assessment models, there has been limited research directed at developing techniques to estimate the parameters used in the models (e.g., natural mortality, selectivity, catchability). Catch and effort data are used to indicate relative changes in lobster abundance in most models. Although various authors have suggested that behaviour of crustaceans can influence what is caught in a trap (Miller 1979, 1990, Addison 1995), Frusher and Hoenig (in press) demonstrated a change in selectivity as the population structure changed. They concluded that this change was brought about by larger lobsters excluding smaller lobsters from entering the traps. This suggests that as the size structure of a population is affected by fishing, the selectivity of the fishing gear changes. Area-specific selectivity estimates are incorporated into the Tasmanian assessment model as a fixed parameter vector that does not allow size-specific selectivity to vary temporally. This paper investigates the implications of changing gear selectivity on the assessment of the fishery.

6.3 Methods

To determine the impact that a change in the selectivity curve could have on interpretation of historic size structure data, we chose a region in northeastern Tasmania where size structure data were available from surveys in 1962, 1963, 1964, 1995, 1996 and 1998 from the same time of year and the same location.

To standardise for different amounts of effort in each of these surveys the catch (number of lobsters caught) was converted to catch rates (number of lobsters caught per trap lift) (Figure 6.1). During each survey, commercial vessels using commercial traps were chartered. The style of trap has not changed over time although escape gaps were introduced between the 1960's and 1990's. To simulate the 1960's fishing

activity, we closed the escape gaps on the fisher's traps that were used in the surveys in the 1990's.

To account for distortions in the size frequency distributions caused by selectivity of the sampling gear, the size composition was adjusted by dividing the standardised number (number per trap lift) in each size class by the corresponding estimate of selectivity.

Frusher and Hoenig (in press) obtained estimates of selectivity from fished populations in southern and eastern Tasmania and a reserve population in southeastern Tasmania (Figure 6.2). Because the reserve population had been protected from exploitation since 1970, there was a large biomass of large lobsters. The size structure of the catch from the reserve was the closest we could find to the relative size structure of the 1960's catch in northeastern Tasmania (Figure 6.3). The reserve size structure is for the first 4 days of sampling as Frusher and Hoenig (in press) found the catch rate of smaller lobsters increased as larger lobsters were removed from the reserve. The sampling regime used during the reserve survey involved removing lobsters to an adjacent aquarium facility as they were caught. These were released back into the reserve at the end of the survey. As large lobsters dominated the earlier catches in the reserve, the size structure for the first four days is closest to the size structure in the 1960's. Thus we use selectivity estimates from the first four days of sampling in the reserve to adjust the size structure of the survey catch in the 1960's.

The size structures in the surveys in northeastern Tasmania in the 1990's most closely resemble the size structure from surveys on the east coast (Figure 6.4). Thus we use selectivity from the east coast to adjust the size structure of the survey catch in the 1990's. A noticeable difference between the east coast size structure and the 1990's size structure from northeastern Tasmania is the number of lobsters greater than 110 mm CL (Figure 6.4). To account for this change and to test the sensitivity of the estimated relative population structure to variation in the selectivity curve, we simulated three selectivity curves that had reduced selectivity of the smaller lobsters and reached peak selectivity at a larger size class (Figure 6.5). The RS curve represents the reserve selectivity curve after it has been smoothed to remove the dips and crests in the curve.

Smoothing was undertaken by fixing the 0 and 1 at the start and end respectively and then taking a running average of 3 for the remaining data points. Dips and crests were still apparent in the curve so this process was repeated on the data from the first smoothing and again on the data from the second smoothing to achieve the RS curve.

The EC curve represents the east coast selectivity which remains unadjusted and the RS-1 and RS-2 curves fit between the RS and EC curves and are simulated to have a peak selectivity 20 and 40 mmCL less than the RS curve respectively.

6.4 Results and Discussion

The minimum legal size limit for males has been 110 mm carapace length (CL) in Tasmania since 1957. In the 1960's, lobsters ≥ 125 mm CL comprised a substantial component of the legal-sized male catch in northeastern Tasmania (Figure 6.1). By 1995 and 1996, the legal sized catch was primarily comprised of new recruits between 110 mm CL and 135 mm CL. In 1998 the legal-sized catch still comprised smaller lobsters but there was a substantial recruitment pulse of sublegal lobsters. This recruitment pulse corresponds to the highest index of puerulus (larval) settlement recorded for the fishery since puerulus monitoring commenced in 1991 (Gardner et al., in press).

When the same selectivity curve (e.g., the reserve selectivity curve) is applied to all size structure data (Figure 6.6 a) the larger lobster biomass seen in 1960's is diminished and the 1998 recruitment pulse increased in relative magnitude. When we apply the east coast selectivity curve to the 1990's data the magnitude of pre-recruits was substantially greater in the 1960's compared to the 1990's (Figure 6.6 b). While relative abundance of pre-recruits is still greater in 1998 than in 1995 and 1996, the recruitment pulse could not account for the abundance of pre-recruits found in the 1960's. Caution needs to be used in interpreting the magnitude of the pre-recruit peak in the 1960's as it relates to the 105 mmCL size class which has resulted from a dip in the reserve selectivity curve (a low selectivity value means a large adjustment to catch rate data). Whenever the estimated selectivity is low, the uncertainty in the adjustment is high. For example, suppose the selectivity for the 105 mm size class is known perfectly to be 0.05. Then the adjustment would consist of multiplying the catch rate for animals in the 105 mm size class by 20, and the standard error for the result would be 20 times the standard error of the catch rate. However, even if we discount the

results for the 105 mm size class as being possible an anomaly, the results in Figure 6.6 clearly indicate higher recruitment in the 1960s than in the 1990s when selectivity curves reflecting the size composition are applied to the data.

To study in a systematic way the effect of changing selectivity on the interpretation of size-specific catch rates we smoothed the reserve selectivity curve to remove the dips and crests and simulated two selectivity curves that fit between the east coast and smoothed reserve selectivity curves. The RS curve was applied to the 1960s catch data, and each of the four curves were applied to the 1990's data (Figure 6.7).

As the selectivity curve moves from the reserve curve to the east coast curve there is a substantial change in the perceived relative abundance. The 1990's data dominate when the reserve selectivity is applied to all years. However, as the selectivity curve moves towards the east coast selectivity pattern, the relative abundance of pre-recruit lobsters in the 1960's becomes dominant. Thus changes in the selectivity curves can lead to substantial changes in population abundance estimates. The application of selectivity curves that account for a change in selectivity as the size structure of the population changes indicates that pre-recruit biomass was substantially higher in the 1960's than the 1990's.

Concomitant with the decline in large male lobsters in north east Tasmania there has been a decline in large female lobster catches (Figure 6.8). This reduction in legal-sized female lobsters has resulted in egg production being less than 10% of an unharvested population (Frusher and Gardner 1999). We suggest that the application of selectivity curves that account for changes in selectivity due to large lobsters reducing the catchability of smaller lobsters, as demonstrated by Frusher and Hoenig (in press), has resulted in a decline in the estimate of the productivity of the Tasmanian rock lobster fishery.

We conclude that gear selectivity is an important component of lobster assessments and greater attention should be given to obtaining accurate selectivity curves. To correctly interpret changes in recruitment associated with the historical exploitation of legal sized biomass, selectivity needs to be obtained for populations of lobsters from currently fished regions as well as regions where the size structure most closely resembles historical size structure or from an unfished population (eg., reserves).

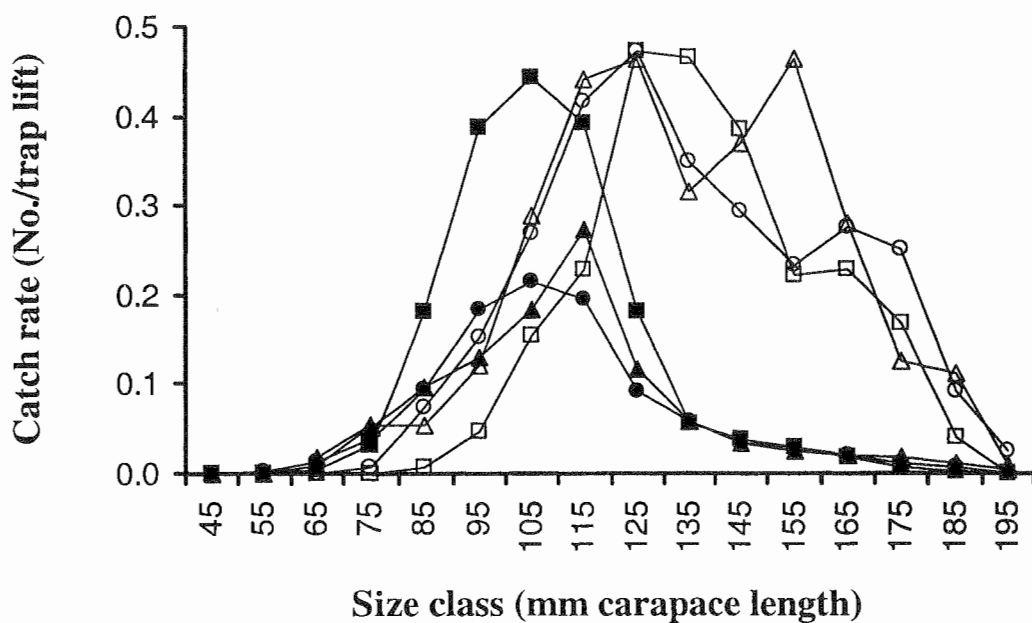


Figure 6.1. Size structure of male lobster catch (unadjusted for selectivity) from catch sampling surveys undertaken in northeast Tasmania in 1962 (open square), 1963 (open circle), 1964 (open triangle), 1995 (solid circle), 1996 (solid triangle) and 1998 (solid square).

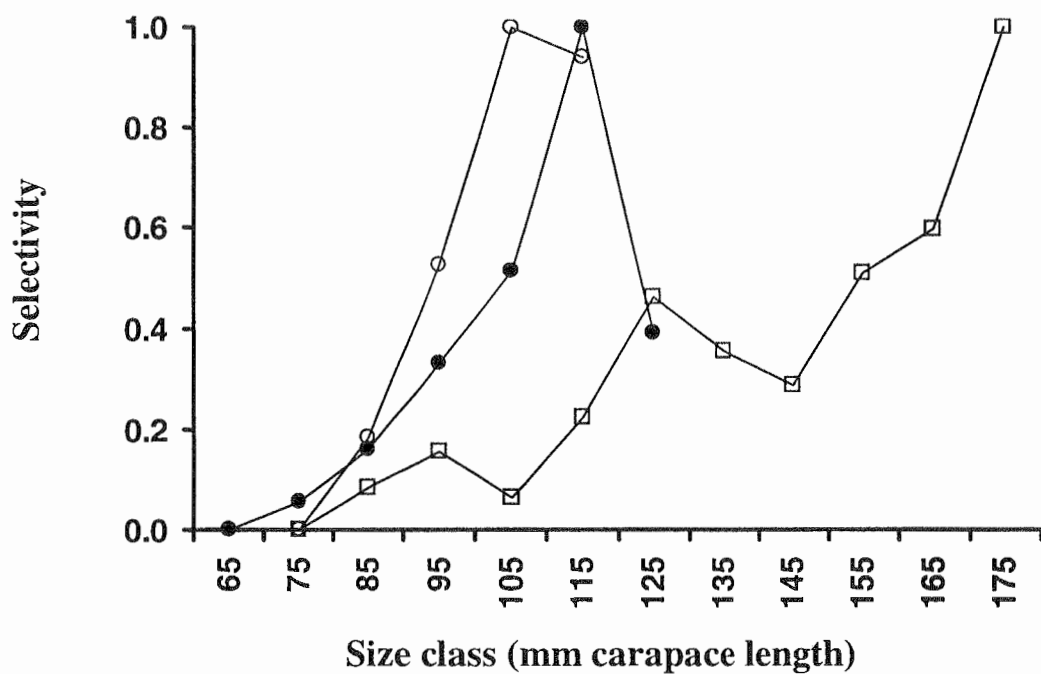


Figure 6.2. Selectivity of male lobsters obtained from four days of research sampling from a reserve (open squares) in southeast Tasmania and from regions of the commercial fishery in southern (open circle) and eastern (solid circle) Tasmania.

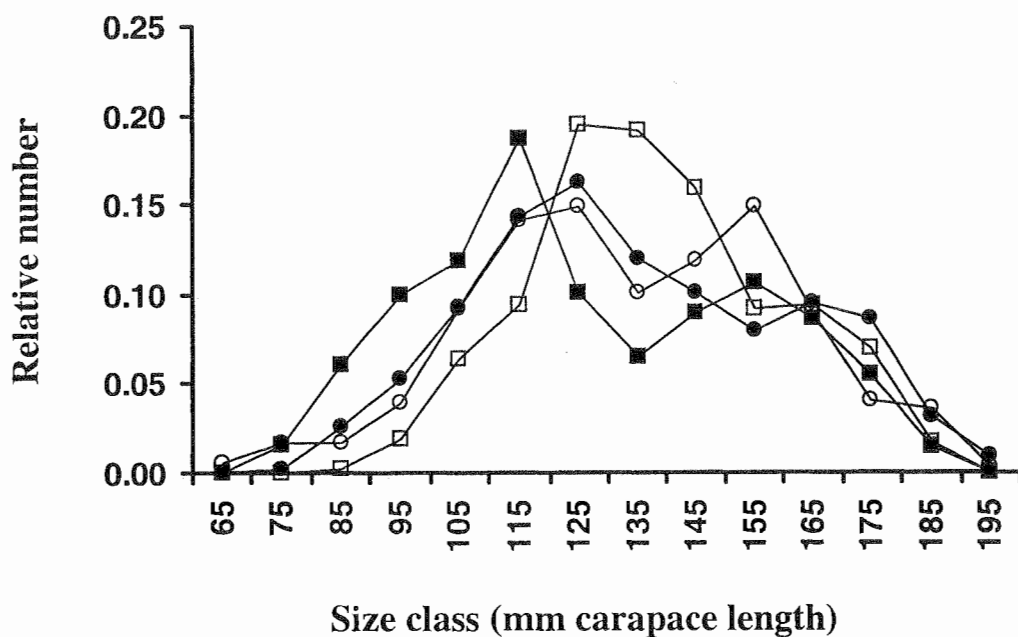


Figure 6.3. Comparison of relative size structure of male lobsters caught in the reserve (first four days of survey) in 1999 (solid square) and in the commercial fishery in northeast Tasmania in 1962 (open square), 1963 (closed circle) and 1964 (open circle). Relative number is obtained by dividing the number caught in each size class by the total number caught in all size classes.

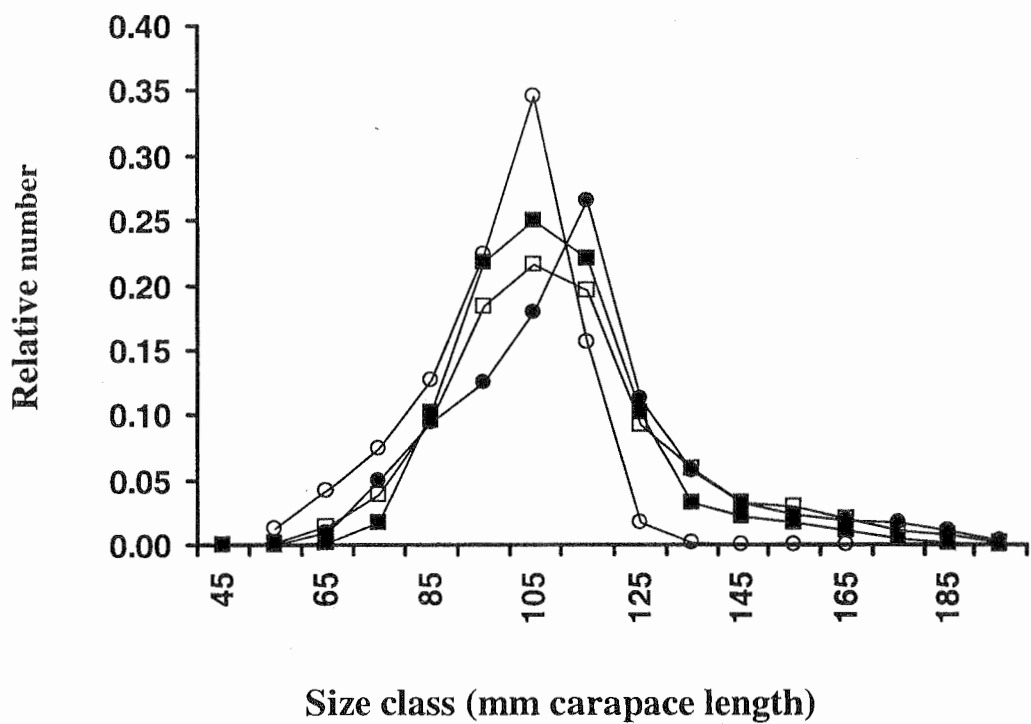


Figure 6.4. Comparison of relative size structure of male lobsters caught on the East coast from 1992 to 1999 (open circle) and in northeast Tasmania in 1995 (open square), 1996 (solid circle) and 1998 (solid square). Relative number is obtained by dividing the number caught in each size class by the total number caught in all size classes.

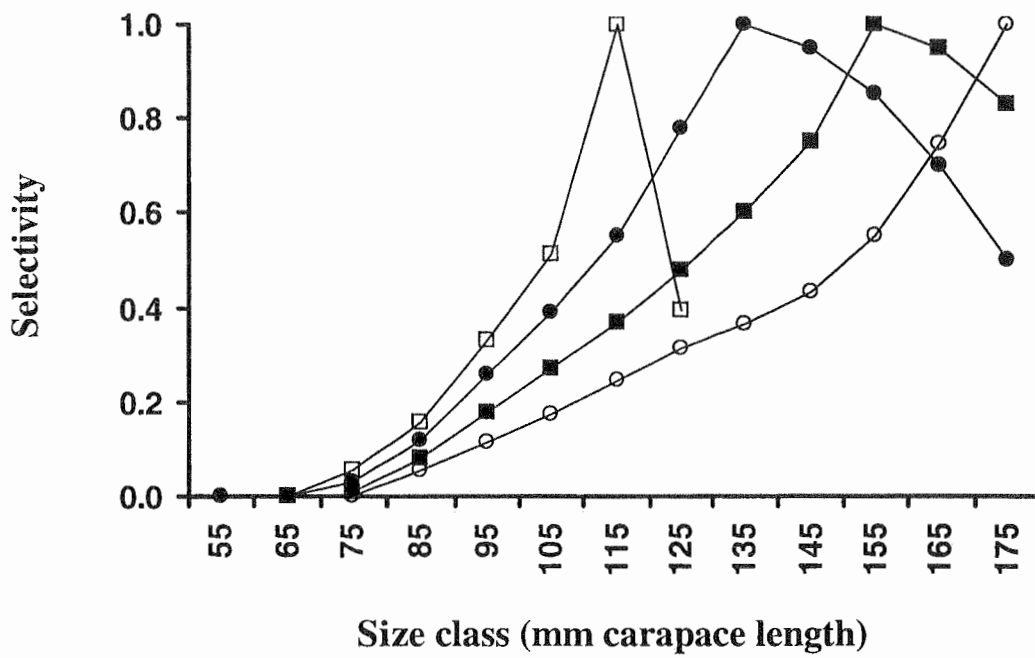


Figure 6.5. Selectivity curves of the reserve (RS [open circle]) after smoothing and east coast (EC [open square]). Simulated selectivity curves (RS-1 [solid square] and RS-2 [solid circle]) have been created to peak 20 mm CL and 40 mm carapace length less than the reserve selectivity curve respectively.

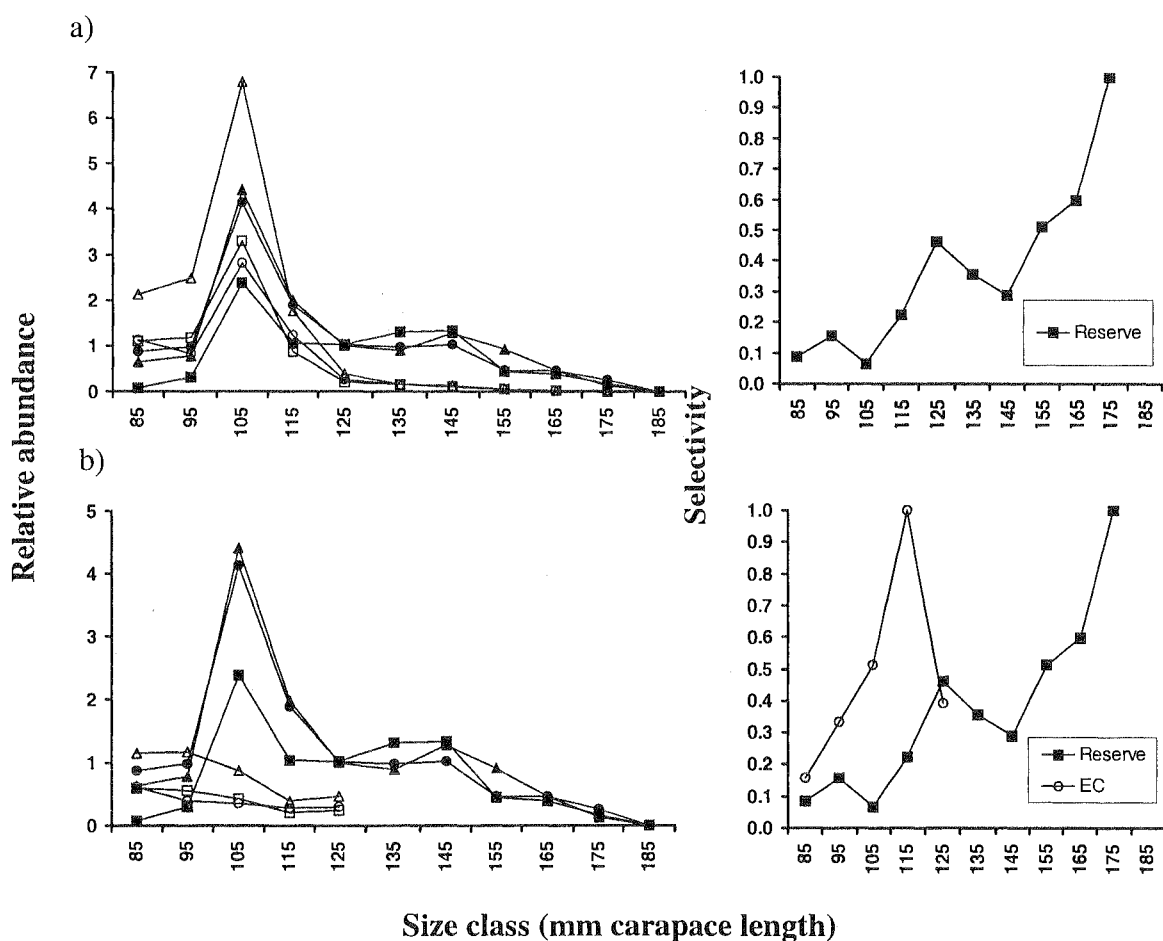


Figure 6.6. Relative abundance estimates (number per trap lift / selectivity) from northeast Tasmania in 1962 (solid square), 1963 (solid circle), 1964 (solid triangle), 1995 (open square), 1996 (open circle) and 1998 (open triangle) using a) the selectivity curve from the reserve (Res) population for all years, and b) the selectivity curve from the reserve population for the 1962, 1963 and 1964 surveys and the selectivity curve from the East Coast (EC) for 1995, 1996 and 1998 surveys.

Selectivity curves are shown to the right of each plot. Unadjusted data are shown in Figure 6.1.

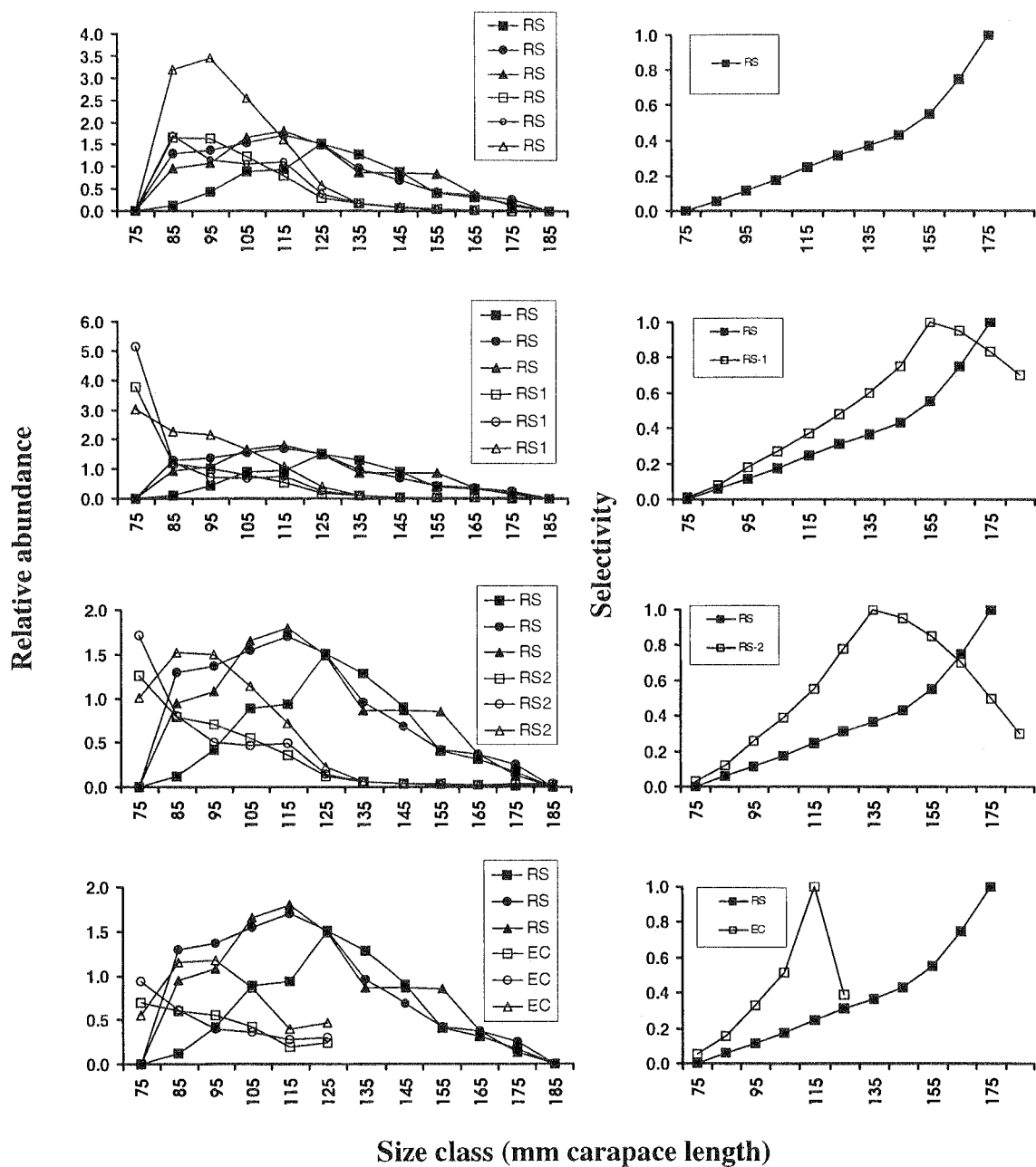


Figure 6.7. Relative abundance estimates from northeast Tasmania in 1962 (solid square), 1963 (solid circle), 1964 (solid triangle), 1995 (open square), 1996 (open circle) and 1998 (open triangle) obtained by applying a smoothed selectivity curve from the reserve surveys (RS), the east coast (EC) selectivity curve, and two simulated curves. The selectivity curves are shown to the right of each plot.

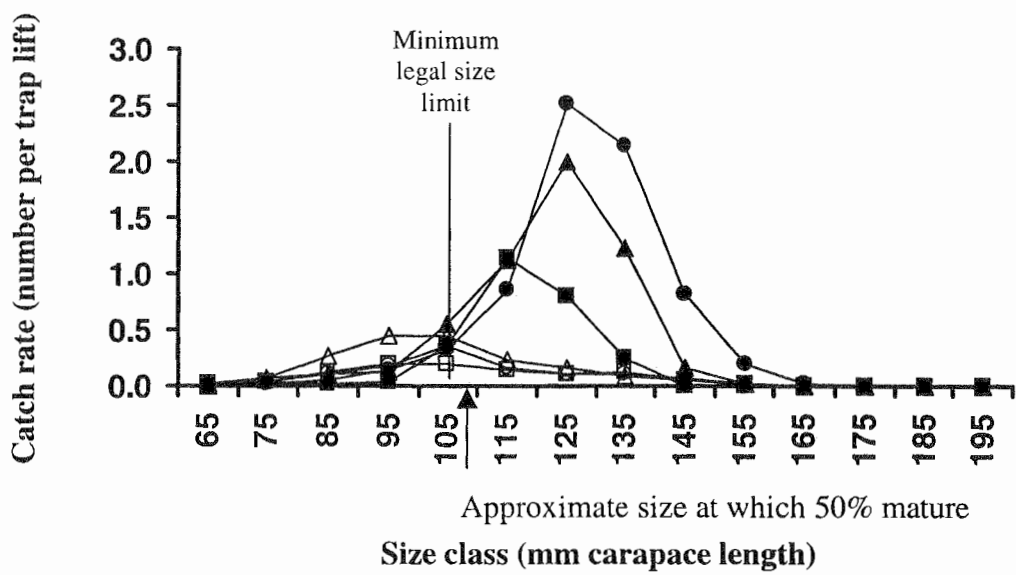


Figure 6.8. Size specific catch rates of female lobsters caught in northeastern Tasmania in 1962 (solid square), 1963 (solid circle), 1964 (solid triangle), 1995 (open square), 1996 (open circle) and 1998 (open triangle).

7. Discussion

This thesis has focused on two important parameters used in the assessments of fish stocks: exploitation rate and selectivity. Knowledge of exploitation rates is crucial if catches are to be related to commercial, or legal sized, biomass. Changes in catch can be caused by changes in recruitment or changes in fishing effort. Thus catches can remain static despite an increasing or decreasing biomass. Catch rates are often used as an indicator of relative abundance under the assumption that the efficiency and selectivity of the fishing gear is constant over time for a given region. Attempts to validate these assumptions are uncommon.

7.1 Exploitation rates.

Frusher et al. (1997, 1998) tested the change-in-ratio (CIR) and index-removal (IR) techniques to estimate exploitation rate in the Tasmanian rock lobster fishery. They found that the single pre-season moult and the minimal movements of lobsters in southern regions of Tasmania made this stock an ideal candidate for testing these techniques. Because of the biology of the lobster, the assumptions concerning recruitment, immigration and emigration are largely met. However there has been uncertainty regarding the assumption of catchability.

Since the work of Frusher et al. (1997, 1998), exploitation rates have been obtained for three to four more years. While the estimates generally appear reasonable, there are times when the methods fail or give contrasting results.

In chapter 2 we developed simple diagnostic tests to determine the suitability of the data for estimating exploitation rate using the CIR and IR estimators. These tests primarily focus on the assumption of catchability. The IR estimator requires catchability to be similar among surveys. In contrast, catchability can vary between surveys for the CIR estimator; however relative catchability of the two size components (size groupings) used in the study is assumed to be constant for the two surveys.

Application of the tests show that catchability does vary substantially between surveys and that the IR estimator seldom produces reliable results. To minimise the

probability of a change in the ratio of catchability of the two size groupings used in the CIR estimator, only small size groupings on either side of the minimum legal size limit are used. In general, the assumption of a constant ratio of catchability appears to be met for the CIR estimator most of the time. Examination of the residuals obtained from the diagnostic tests provided additional information on the magnitude of catchability change and also indicated periods of recruitment to the fishery.

Unlike in southern regions of the fishery, in northern regions recruitment is known to occur within the fishing year. This invalidates use of the CIR and IR exploitation rate estimators. Exploitation rate pertains to the fate of those lobsters that were legal-size at the start of the fishing season. This can be determined by following the fate of tagged lobsters that were legal at the start of the fishing season. Recent developments in multi-year tagging models appeared suited for northern regions of the fishery. From 1992 to 1995 three research surveys were undertaken each fishing year in northwestern Tasmania. During these surveys legal sized lobsters were tagged to determine growth and migrations. Although these surveys were not designed for estimation of exploitation rate, the data appeared useful for investigating the use of multi-year tagging models. In Chapter 3, models were developed that incorporated the three tagging surveys each fishing year. The preferred model was an adaptation of two previously described models: a model with two tagging events per year that had previously only been tested on simulated data (Hearn et al. 1998) and a model used to describe an apportioning of mortality among periods of the year (Hoenig et al. 1998). The model provided relatively precise estimates of annual exploitation rates (fishing mortality) which were similar to those obtained from a spatially-explicit size-structured assessment model (Punt and Kennedy 1997).

In addition to exploitation rates, the model provided estimates of tag reporting rate and natural mortality. Estimates of tag reporting rate were precise and indicated that few fishers were cooperating with the tagging project. Natural mortality estimates were extremely low and precision was very poor.

Using the preferred model as a template, simulations based on the estimates of fishing and natural mortality from the preferred model were undertaken to investigate ways of improving the precision of natural mortality estimates (Chapter 4). Simulations indicated that the best improvements in natural mortality estimates would be obtained by increasing the duration of the study (3 to 6 years) rather than increasing the number

of tagging events per year or the number of lobsters tagged per tagging event. Substantial improvements could also be obtained by improving tag reporting rate. While it is recognised that rewards and other incentives have proven useful in other tagging studies, there is no guarantee that tag reporting rate will increase and be maintained if reward levels are increased.

In Tasmania, T-bar tags have been used and these have been inserted ventrally because tag loss was considered to be lower than if tagged animals were tagged dorsally (Treble 1996). However, reporting of these tags is considered lower as fishers have to invert the lobster to look for the tag. Although dorsal tagging may have a slightly higher tag loss rate, improvements in tag reporting rate may offset tag loss.

7.2 Selectivity

Analysis of tag recapture data also provided information on selectivity of the fishing gear. Daily removals of lobsters in a reserve population were undertaken to simulate fishing activity (although all lobsters were removed rather than only legal sized lobsters). Large lobsters were found to dominate the initial catches and as these were removed the catch rate of smaller lobsters increased. Thus the fishing gear appears to select for large lobsters but as these are depleted the selectivity shifts towards smaller lobsters.

In support of the effect that large lobsters may have on the catch of small lobsters, we compared the number of small and large lobsters caught in individual traps in three regions of the fishery that had substantial differences in size distributions. Negative correlations were found between the number of small and large lobsters caught in traps and these correlations were stronger in populations that had a more extensive size distribution of large (legal-sized) lobsters. The extent of the size distribution of large lobsters in Tasmania is primarily determined by their growth rates. Growth rates increase from southern to northern regions of Tasmania (Frusher 1997). Thus the number of lobsters in the larger size classes increases from southern to northern Tasmania. Concomitant with the increase in the size distribution of legal sized lobster from southern to northern Tasmania, there were larger negative correlations in the North than in the South.

Finally we analysed tag and recapture data which were obtained on the south and east coasts of Tasmania in regions of the commercial fishery to determine selectivity.

Tagging was a component of catch sampling surveys which were undertaken at the start, middle and end of each fishing season from the 1992/ 1993 to the 1997/1998 fishing seasons. Again, this tagging project was undertaken to obtain growth and movement data and few sites were sampled twice during each survey. Only recaptures obtained from tagging during the same survey could be used to estimate selectivity. Commercial fishing between surveys resulted in the size structure of the legal-sized portion of the population changing between surveys during a fishing year. Although the data sets were not ideal, it was possible to obtain estimates of selectivity although this required data from years and sites within regions to be aggregated.

In southern regions of Tasmania, the fishery is primarily based on recent recruits due to the high exploitation. This, combined with the slow growth rates has resulted in few lobsters greater than 5 mm CL above the legal size limit being present in the fishery. No difference between selectivity curves for the start, middle and end of fishing season surveys were found for southern Tasmania. In contrast, in eastern regions where growth rates are faster, there is a larger size ranges of lobsters above the legal size limit. Differences between the selectivity curves for the start, middle and end of year fishing surveys were discernible in eastern Tasmania. The most substantial change was seen between the start and middle of year surveys for both males and females and this corresponded to the period when approximately 80% of lobsters are harvested. Between the middle and end of year surveys, male selectivity curves showed limited change as the remaining lobsters were caught. The female selectivity curve for the end of season survey was similar to the curve for the start of season survey. This would be expected as females undertake their annual moult after the middle of season survey and are protected from being retained. Thus, unlike males, the female size distribution at the end of the season should reflect the size distribution obtained at the beginning of the next season. Thus selectivity of the fishing gear can change within a fishing season as larger lobsters are removed from the population by harvesting.

Fishing gear selectivity vectors are an important parameter (either explicitly or implicitly) in the assessment of most fisheries. They are used to relate the catch obtained from a specific fishing gear to the population. Change in population structure can be obtained by direct comparison of catch rate data from different surveys. More sophisticated analyses apply selectivity curves to the raw survey catch data to obtain a

more 'accurate' representation of the population structure. Although some assessment models account for spatial changes in selectivity, temporal change is not accounted for in any of the models of which we are aware.

The mechanism that accounts for the change in the selectivity of the fishing gear as large lobsters are removed from the population, as described in Chapter 5, is uncertain. Size related dominance hierarchies have been suggested by other authors (Miller 1995, Addison 1995) and would appear to be a reasonable explanation. The impact that altered selectivity curves can have on interpretation of catch and effort data is described in chapter 6.

Size structured catch rate data were available from surveys undertaken in the 1960's and 1990's in northern Tasmania. Between the 1960's and the 1990's there was a substantial decline in the legal sized biomass as fishing effort increased in this region. As selectivity curves for northeastern Tasmania are unavailable, we used selectivity curves that were available from other regions that had a similar size range of lobsters caught in traps. Although the selectivity curves used were those available that best matched the size structure of the northern populations, there were minor differences between the size structure of the northern populations and the populations where the selectivity curves were derived. To account for these differences, and to determine the sensitivity to change in the population size structure, we adjusted the population structure by using a series of simulated selectivity curves.

Adjusting the size structured catch data, by the selectivity curve for the population that most closely resembled the size range of lobsters caught, resulted in a substantial change in the estimate of the proportion of small lobsters in the fishery. This change was not apparent by either analysing the catch rate data unadjusted, or adjusted by the same selectivity curve for all years. The resultant size distributions showed that the undersized population of lobsters has declined substantially with the decline in legal sized biomass. The extent to which this decline has occurred will be dependent on the use of accurate selectivity curves. Simulated selectivity curves that accounted for different degrees of size dominated hierarchical effects demonstrated that perception of the undersized component of the lobster population is sensitive to small changes in the selectivity curves used to adjust the size specific catch data to relative abundance. Thus it is suggested that the interaction of larger lobsters with small lobsters has led to misrepresentation of the change in undersized portions of lobster populations found in

catches from the fishery. It is possible that the estimated decline in egg production in northeastern Tasmania to less than 10% of its unfished egg production has led to a decline in recruitment. However, such a decline would go unnoticed if the relative populations were not adjusted to account for behavioural interactions such as large lobsters influencing the catch rate of small lobsters.

7.3 Relevance of results to the current Tasmanian rock lobster fishery

In 1998 an individual transferable quota system was implemented in the Tasmanian rock lobster fishery. An objective of this new management regime was to allow the biomass of legal sized lobsters to increase (Anon 1997). The rebuilding of the biomass is likely to affect the methods currently used to estimate exploitation rate and biomass. A rebuilding biomass will be associated with lower exploitation rates. Both the CIR and IR methods have greater precision when exploitation rate is high (Paulik and Robson 1969, Frusher et al. 1997). In contrast, tagging models may remain effective in estimating exploitation rate as the biomass rebuilds. While the lower exploitation rates (and fishing mortalities) may decrease the precision of fishing mortality estimates as fewer returns are achieved each recovery period, tagged animals will remain longer in the fishery and thus contribute tag returns over several years after tagging. For estimates of parameters that are held constant over several years, such as natural mortality and tag reporting rate, the precision of these estimates should improve, as they would be linked to several different years of tagging. The increasing biomass would also improve catch rates that would enable greater numbers of legal sized lobsters to be tagged for a set period of research effort and thus improve the cost effectiveness of this method.

The rebuilding biomass will result in an increased number of larger sized lobsters that will be expected to influence the catch rate of smaller lobsters. If selectivity curves, which do not account for the behavioural response of lobsters, are used to adjust size specific frequency distributions for sampling bias, the initial catches of smaller lobsters may suggest recruitment decline as they are suppressed from entering traps by larger lobsters. Accurate selectivity curves that represent the size structure of the population being examined are required to determine true population size structure. Regular estimates of selectivity will be required as the size structure of the population changes.

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